From Impact to Resource – Case studies of Bioenergy, Biomaterials and Associated Carbon for Climate Change Mitigation

Von Auswirkung zu Ressource – Fallstudien zu Bioenergie, Biomaterialien und assoziiertem Kohlenstoff im Kontext der Klimawandelminderung

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Erklärung:

Hiermit erkläre ich, die Dissertation selbstständig und nur unter Verwendung der angegebenen Hilfen und Hilfsmittel angefertigt zu haben.

Declaration:

I hereby declare that I completed the doctoral thesis independently based on the stated resources and aids.

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"Future historians may note that although humanity solved one unexpected environmental problem, it bequeathed many more through its failure to take a holistic approach to the environment."

(Shanklin 2010)

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Zusammenfassung

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Eine verstärkte Nutzung von Energie und Rohstoffen auf Basis von Biomasse ('Bioenergie', Biomaterialien') gilt als wichtiger Beitrag, um den anthropogen begründeten Klimawandel zu mindern. Die Vorzüglichkeit von Bioenergie/-materialien begründet sich darin, dass sie im Vergleich zu ihren fossilen Referenztechnologien pro bereitgestellter Energie-/Produkteinheit weniger klimawirksame Gase (CO_2, N_2O, CH_4) emittieren. Die Treibhausgase (THG), die entlang einer Bereitstellungskette frei werden, werden in Bilanzen aufsummiert und auf eine gemeinsame Einheit (‚CO_2-Äquivalente') skaliert. Die Bilanzerstellung erfordert zahlreiche quantitative Informationen, die mit unterschiedlichem Genauigkeitsgrad vorliegen, so dass auch die gesamte THG-Minderung gegenüber der Referenz mit einer gewissen Unsicherheit behaftet ist. Frühere Studien haben ergeben, dass diese Unsicherheiten, oder auch eine Nichtberücksichtigung von Effekten entlang der Bereitstellungskette, die Bilanz ins Negative umschlagen lassen kann, die Biomassenutzung also nicht mehr vorzüglich wäre.

Die vorliegende Arbeit analysiert, wie diese Unsicherheiten die Aussage über Vorzüglichkeit von Biomassenutzung im direkten Vergleich zu einer Referenz, aber auch in komplexeren wirtschaftlichen Nutzungsketten, beeinflussen. In der ersten Fallstudie wurde mit der stationären Biostromerzeugung aus der Vergasung von Pappel-Hackschnitzeln aus Kurzumtrieb eine Verfahrenskette mittels Stoffstromnetz modelliert und analysiert, für die Emissionsdaten aus Felduntersuchungen zur Verfügung standen. Mittels Monte-Carlo-Analysen wurde gezeigt, dass diese Biostrombereitstellung auch unter Berücksichtigung der Unsicherheiten weniger Klimagase emittiert bzw. sogar zusätzliches CO_2 sequestrieren könnte. Den größten Beitrag zur Gesamtunsicherheit der Vermeidungsleistung hatte die Unsicherheit des Bodens als Kohlenstoff-Senke (76 %), gefolgt von der des Ertrags (15 %), sowie vermiedenen Referenz-N_2O-Emissionen (2 %). Letztere sind bisher nicht Standard-Bestandteil in Vermeidungsanalysen, waren allerdings in der Fallstudie vergleichbar bedeutsam wie Wärmeallokation und Wirkungsgrad (je 1%).

Die zweite Fallstudie widmete sich einer Biomassenutzung in komplexerer Konstellation. Der aktuellen Forderung folgend, Biomassenutzung mittels mehrerer Kriterien zu bewerten, bezog diese Studie neben THG-Emissionen sowohl den Bedarf an Agrarfläche als auch an fossilen Ressourcen (Erdöl, Erdgas) mit ein. Es galt zu bestimmen, wie bei gleichzeitiger Verfolgung des Klimaschutzziels diese begrenzten Ressourcen bei der Dämmung von Häusern am effizientesten eingesetzt werden sollten. Agrarfläche diente entweder zur Bereitstellung von Biomaterial (Hanffasern) oder Bioenergieträgern (Pappelhackschnitzel, Maissilage). Komplementär wurden fossile Rohstoffe entweder zur Material- oder Energieerzeugung eingesetzt. Aus den drei Kriterien konnte keine eindeutige Vorzugslösung abgeleitet werden, da auch die untersuchten Szenarien kein einheitliches Bild ergaben. Effiziente Recyclingsysteme vorausgesetzt, könnte eine stoffliche der energetischen Nutzung von fossilen Rohstoffen vorzuziehen sein, und Agrarflächen eher der Bioenergiebereitstellung dienen.

Klimagase können ebenso wie Land und fossile Rohstoffe als begrenzte Ressource aufgefasst werden, da die Aufnahmekapazität der Erdatmosphäre begrenzt ist, wenn keine bzw. nur moderate Klimaänderungen erfolgen sollen. Basierend auf dem Produktivitätskonzept, das gängige Praxis für die Bewertung von Ressourceneffizienz ist, wurde im dritten Beitrag ein Vorschlag entwickelt, wie die Nutzungseffizienz des in der Biomasse bereitgestellten Kohlenstoffs bewertet werden könnte. Dieser *CUDe* genannte Ansatz wurde auf Verfahrensketten der Biogaserzeugung aus Maissilage und auf Hanfdämmung exemplarisch angewendet und diskutiert.

Kohlenstoff nicht nur hinsichtlich seiner (Klima-)Auswirkungen zu bewerten, sondern ihn vor allem als Ressource zu betrachten, die es möglichst effizient zu nutzen gilt, könnte es ermöglichen, die Optimierungspotentiale zu erschließen, die entlang von Biomassenutzungsketten existieren, und dabei gemeinsam mit Klimaschutz weitere aktuelle Handlungsfelder zu adressieren.

Die Dissertation schließt mit einer Aufstellung offener Fragen.

Extended Summary

Introduction

Biomass-based energy (bioenergy) and materials (biomaterials) are considered an important contribution to the mitigation of human-induced climate change and as relevant feedstock in a future economic system (bioeconomy). Related policies that target the future bioenergy share in energy mixes are in place on the national, European as well as on the international level (Climate Change Package; European Parliament 2008).

The underlying assumed preferability of bioenergy and biomaterials when compared to conventional, fossil reference technologies is based on lower greenhouse gas (GHG) emissions per energy or product unit. This preferability is estimated from mitigation analyses which balance GHG that are emitted along the transformation chain (mainly carbon dioxide/CO_2, nitrous oxide/N_2O, methane/CH_4), and express their total amount per unit of generated energy with a common unit (CO_2 equivalents). Quantitative information on sub-processes is necessary to calculate such balances which is available only at different levels of certainty. This implies that the achievable climate change mitigation is also associated with uncertainty. Despite methodological recommendations for mitigation analyses, previous studies have shown that uncertainties as well as the omission of effects along the biomass transformation chain can result in contrary results, that is, biomass usage may not yield any mitigation effects.

Objectives

Against that background, this publication-based dissertation aims to contribute to the discussion about the reliability of CC mitigation assessment of biomass application in an increasingly bio-based, low-carbon economy that also fulfils sustainability constraints of resource conservation. Using case studies (6.1-6.3), it aims to answer the following questions:

- How much CC mitigation can we expect from bioenergy and how reliable are any mitigation potentials–if they exist?
- What are appropriate baselines against which GHG fluxes are balanced in the agricultural context? Is there a need to include more processes into the assessment?
- How relevant are uncertainties in the bioenergy process level if biomass usage is assessed in a broader context as in bioeconomy or in multi-criteria assessments?
- Do alternatives to mitigation analyses exist in order to address agriculture-specific characteristics of biomass generation and transformation?

Approach

The analyses used a life-cycle-based approach (see 4.1). They included the relevant production processes of crop cultivation and its production factors (fertilizers, pesticides, fuels, seeds) which emit the three most relevant GHGs in an agricultural context: N_2O, CH_4 and CO_2. The bioenergy pathway in the first case study (Hansen *et al.* 2013) was modeled using a Life Cycle Assessment Tool (Umberto® 5.6) (ifu&ifeu 1994-2011) which includes a function to address uncertainty in the material flow models by Monte Carlo analyses (4.2.2). Such analyses are being increasingly performed in LCA studies of biomass usage (Table 4.4). Varied parameters were: Soil N_2O emissions from unfertilized poplar plantations and unfertilized rye plots on sandy soils; allocation of heat extraction; transport distance; electric conversion efficiency; wood chips yield; soil organic carbon change (sink); reference electricity generation in Germany; and global warming potentials (GWP_{100}) of N_2O and CH_4 (see Table 6.6). An extended land use change (LUC) assessment as a variation of the usual balancing approach was included in Hansen *et al.* (2013). On the one hand, absolute N_2O emissions from unfertilized[1] poplar SRC plots were considered. Additionally, these emissions where balanced against the N_2O that would

[1] No N-containing fertilizer, but phosphorus and potassium fertilizer

have been emitted from the reference crop rye. The reference energy system was taken from Klobasa *et al.* (2009) because their model very specifically assesses the substitution effects of biomass in the electricity mix (4.1). Allocation was avoided through system expansion where possible. Otherwise, a mixed approach was followed as in Eady *et al.* (2012).

In Hansen *et al.* (2016a), the results of the Umberto® model were merged in spreadsheets (MS Professional Plus 2010) with additional data from the different unit processes to represent two strategies for the insulation of buildings. For these strategies, a scenario analysis was performed due to the complexity of systems and resulting problems in safeguarding the independence of parameters for a Monte Carlo analysis. Additional data were taken from common LCA data repositories as well as from qualitative telephone interviews with stakeholders. In addition to climate change impact, this study also analyzed resource-related indicators for fossil fuels (crude oil, natural gas) and agricultural land in a multi-criteria approach.

In distinction to climate change mitigation as an impact-oriented assessment of biomass usage, a five-plus-one step approach was developed in Hansen *et al.* (2016b; Carbon Utilization Degree/*CUDe*). Following a process chain assessment (Figure 6.8), the *CUDe* approach aims at identifying how efficiently carbon (C) is used in biomass utilization chains. The ratio of the overall productive carbon to the carbon that was originally available in the biomass was defined as Carbon Utilization Degree (*CUDe*).

Results

The first case study (Hansen *et al.* 2013) (6.1) modeled and analyzed a transformation chain where poplar wood chips (SRC) are gasified for stationary electricity generation. N_2O emission data were available from trial SRC sites.

Monte Carlo analysis results indicated that the bioelectricity thus produced could contribute to mitigation with high agreement/medium evidence (274±21 g CO_{2e} per MJ electricity generated)(6.1.3), subject to the condition that site and management conditions are well known and that soil-bound N_2O emissions are low. The relative mitigation potential of this electricity would be approximately MP_B=114±8 %, with the value greater than 100 % denoting a moderate overcompensation of emissions, i.e. sequestration. The inclusion of parameter uncertainties and uncertainties in the model structure resulted in a comparatively low relative variability (8 %) for this calculated mitigation strategy.

The most important contributors to uncertainty were soil organic carbon stock increases (76 %), wood chips' yield (15 %), mitigated N_2O emissions compared to the reference crop rye (*Secale cereale* L.) (2 %). The latter are not a standard component in mitigation analyses thus far; nevertheless, they were of comparable importance for the total uncertainty as heat allocation and conversion efficiency (1 % each). Modelling decisions had a strong influence on the relative importance of the individual parameters, but a relatively low impact on the overall mitigation effect.

One relevant characteristic of the biomass feedstock used for this energy generation pathway was its zero-fertilization preference, as well as the opportunity that it provided to increase soil carbon stocks at the plantation sites (3.1). Whether such preconditions apply to other bioenergy generation pathways must be assessed separately for each. According to a literature review, bioelectricity from SRC could possibly sequester 32 g CO_{2e} MJ^{-1} or emit up to 228 g CO_{2e} MJ^{-1} (Table 7.1 as update of Table 6.7 in Hansen *et al.* (2013)). Higher emissions were usually associated with electricity generation chains with energy-intense sub-processes (drying, pelleting). Compared to other bioenergy generated from woody biomass, the modelled poplar wood chip gasification was well within in the range of the emission factors, whereas the mitigation effect was somewhat higher. This was due to the comparably high reference emissions of the German case study compared to the natural gas reference technology that was often used in the other studies.

The conversion efficiency of the energy embedded in the biomass is a prerequisite for climate mitigation through bioenergy. It is only achievable if, besides electricity, heat is also used sensitively, for example via Combined Heat and Power. This in turn demands the inclusion of bioenergy usage in a broader economic context. There is a need to understand how important uncertainties in the assessment of bioelectricity remain if electricity generation becomes just one out of many unit processes in a larger system analysis. This is especially true if more assessment criteria than climate change impact are of interest.

The production of insulation materials is an economic activity that demands electricity as well as heat energy. Hence, by addressing a multifaceted problem as for example "What is the preferable resource allocation of *fossil fuels* and *agricultural land* under the constraint of minimum GHG emissions for insulation production", from case study 2 (Hansen *et al.* 2016a) it arose that unit process uncertainties may be less important than system uncertainties from modelling choices.

This second case study (6.2) dealt with the question if mitigation analyses are a helpful tool for decision support if biomass usage occurs in more complex constellations than single technology comparisons. Besides the GHG emissions, the demand for cropland and fossil resources was included as well, in order to decide how restricted resources (cropland, fossil fuels) should be used most efficiently, and to mitigate climate change at the same time. This inclusion of resource usage aspects instead of a pure impact assessment represented the growing awareness of the demand for sustainable resource use strategies.

The case study was based on system[2] definitions that safeguarded that the overall performance of both systems (*biomaterial*/land-based or *bioenergy*/land-based) was identical and that the overall aim of CC mitigation was targeted. Studied systems provided identical insulation effects for buildings, either from natural fibers or polystyrene. Cropland supplied either biomaterials (hemp fibers) or bioenergy carriers for insulation production (wood chips, maize silage). Fossil fuels provided, in turn, production energy or material feedstock. The multi-criteria analysis included several scenarios to account for the wide range of possible co-products and reference systems. From the three indicator results in the system expansion approach, none of the resource usage strategies would be clearly preferred (Table 6.13). However, depending on recycling concepts that are in place, the material usage of fossil resources might be preferable over the energetic one, whereas the resource cropland could provide bioenergy.

In the basic scenarios, both strategies had comparable GHG emissions, whereas the *biomaterial* strategy needed more land but less fossil resources than the *bioenergy* strategy. If recycling was accounted for, a *bioenergy* strategy became more preferable because it seemed to jointly address the goals CC and efficient resource use. Recycling was addressed for the criterion fossil resource demand only, whereas no statement regarding LU in both systems combined with recycling could be made.

The study provided a more detailed picture of how to arrive at decisions which insulation materials should be chosen in a "broader picture", that is, if society aims at addressing several goals jointly. Yet it did not state how societies should provide heating or cooling energy for buildings, as this element of the system was excluded (see definition of functional unit in Hansen *et al.* 2016a). As well, it did not contribute to a discussion if we should insulate buildings or not, as this is already clear: The heating and cooling demand of buildings is responsible for 40 % of energy consumption within the EU (European Parliament and the Council 2010) and 35 % worldwide (IEA 2006). The insulation of buildings could reduce this demand significantly and appreciably contribute to climate change mitigation.

In such complex production chains and networks, it seemed advisable to address several criteria in order to reduce the impact of uncertainties caused by system complexity and to identify trade-offs to other indicators (avoid leakage effects). The occurrence of co-products and their use had a great influence on study results. This, on the one hand, indicated once more that as much as possible of agricultural (co-)products should be put to use. On the other hand, it illustrated that is not advisable to generalize results: Assumptions about the region where agricultural production takes place had a large effect on results and could even result in a change in the ranking of the criteria (see scenario Hemp83_Sunfl in Table 6.13).

[2] In the manuscript, the terminology "strategies" has been used instead of "systems".

Considering the difficulties that multi-criteria assessments face in the agricultural context, it could be helpful to adopt further methodologies. The impact-related indicator *GHG emissions* could also be re-interpreted as a resource indicator: GHGs could be considered a limited resource as well, since the capability of Earth’s atmosphere to act as a sink for GHG gases is limited, at least if climate change should not exceed a threshold. Following this line of thought further, one could define carbon in biomass as a restricted resource which should be used most efficiently. Based on the productivity concept, which is common for the evaluation of resource use efficiency, the third article (Hansen *et al.* 2016b) tentatively applied this option. It presented an approach of how to evaluate the usage efficiency of biomass carbon within biomass transformation chains. This *CUDe* approach was applied to two technologies as examples.

In a generalized case study, the *CUDe* approach indicated the sustainability of using biomass (fiber hemp) for building insulations. The sum of productive carbon was greater than 100 % due to the cascading use of the biomass in this transformation pathway. In another application on the transformation of maize to biogas and its subsequent use for energy generation, *CUDe* indicated some optimization potentials. The implementation of additional CO_2 usage combined with an upgrading process could improve this biomass usage pathway in terms of sustainability of carbon use. Upgrading alone did not improve C productivity; on the contrary, it was reduced due to additional process emissions.

Concluding Remarks

According to the results of the case studies, bioenergy, in particular bioelectricity, could contribute to climate change mitigation efforts with high agreement/medium evidence (terminology according Mastrandrea 2010). Under specific conditions – increasing SOC stocks for instance or/and reduced N_2O emissions relative to reference crop –, it could possibly re-fixate atmospheric carbon to longer-lasting C-pools. The amount of SOC contribution to the mitigation is associated with higher uncertainty due to missing long-term data. The effect of species-related reductions in N_2O emissions would contribute to the mitigation effect with medium to high confidence due to increased evidence from measurements. As a result, biomass cultivation should be baselined against more than the agreed-on (but still uncertain) C stock changes, but should account also for crop-specific differences in N_2O emissions.

However, to reach necessarily ambitious global GHG reduction goals, it would be more important to focus on the demand side potentials and reduce overall energy consumption. Biomass in whatever context needs to be used as efficiently as possible; this includes co-products and in a cascading way. Seeing carbon as a central element in organic compounds and as an indispensable resource for life that should be used most efficiently might help to tap the full optimization potentials along biomass transformation und to conjointly address other problems. Such developments could be seen as the beginning of a paradigm shift where C is no more seen as a threat but as an asset instead.

The thesis closes with a compilation of open questions.

1 Introduction

Uncertain Trajectory of Biomass Production Systems

Human society relies on solar energy. For thousands of years, agricultural societies transformed that solar energy, restricted to land, into essential products (food, feed, materials, energy carriers) (Wackernagel & Beyers 2010; Jering *et al.* 2013), with residues being recycled within the system (Figure 1.1). The evolving agro-industrial system modified that practice fundamentally by using fossilized solar energy in fuels or fertilizers produced with fossil energy. This was accompanied by the system being less dependent on land. On the other hand, carbon dioxide (CO_2) was emitted from that fossil energy use and over time, atmospheric concentrations of this important greenhouse gas (GHG) increased considerably, resulting in additional climate change (CC) to the natural one. Negative effects from that accelerated change are already observable, such as for example an increase in extreme weather events, rising sea levels, and changes in crop growth patterns; all of these are expected to increase even further (IPCC 2014). It needs to be evaluated how future production systems could be framed that account for the limited absorption capacity of the atmosphere and oceans to limit further climate change (Le Quere *et al.* 2009; Jiankun & Mingshan 2011) and as well for the limited land availability, and still provide enough basic products for a growing society.

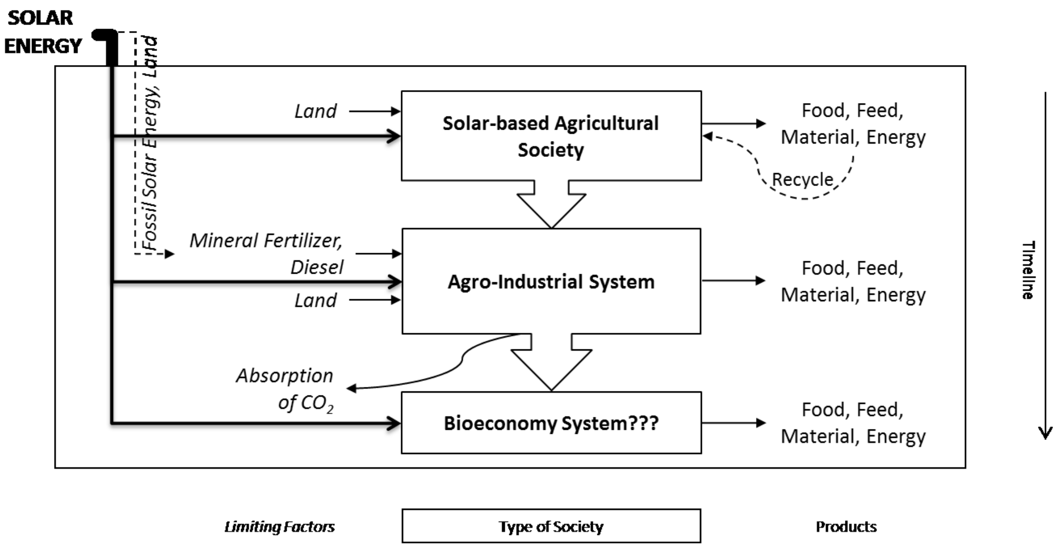

Figure 1.1 Limitation of societal systems by agricultural production factors; from Hansen and Wolf (2015) based on an idea by Wackernagel and Beyers (2010)

Targets Set by Policy

As an effective strategy to mitigate climate change and to maintain fossil resources as well, hope has been placed in a switch from the use of fossil resources to an increased use of biomass for modern forms of bioenergy generation (IEA 2012) as well as for material provision (Koh & Ghazoul 2008; Angelou 2013).

The European Union implemented a corresponding strategy in 2008 (Climate Change Package (European Parliament 2008)) in order to ensure by 2020 a 20 % reduction in GHG emissions, a 20 % improvement in energy efficiency, and a 20 % share for renewables in European energy mix (20/20/20 targets). Similar policies exist worldwide, specifically for biofuels (Sorda *et al.* 2010); ASEAN countries (Mofijur *et al.* 2015); or in the United States' Energy Independence and Security Act (U.S. Congress 2007). In Europe, legal regulations are in force (Renewable Energy Directive, RED (European Parliament and the Council 2009)), and nationally implemented (for instance in Germany: Renewable Energy Sources (RES) Act (BMWI 2014)). The German RES Act set a target of an 80 % renewable electricity share by 2050, in steps of 40-45 % by 2025 and 55-60 % by 2035. Transformation studies stated even more ambitious targets of 100 % renewable energy systems (Denmark (Lund 2009); Germany (SRU 2013)). Bioelectricity has been considered important in a future renewable electricity mix as it may balance load fluctuations which result from variations in sun and wind availability (Mühlenhoff 2013).

In 2015, 32 % of German electricity was generated from renewables, of which 9 % was bioelectricity and 2 % was from solid biomass respectively (BMWI 2015).

Criticism of Climate Change Mitigation through Bioenergy

How much both bioenergy and increased biomass usage really contribute to CC mitigation and resource conservation under sustainability constraints has been and is still being questioned (WBGU 2009). Biofuel generation draws numerous criticisms. These concern possible land use change (LUC) effects (Searchinger *et al.* 2008; Searchinger *et al.* 2009))(see later in 4.1.5); N_2O (nitrous oxide) emissions from fertilizer application in bioenergy crops (Crutzen *et al.* 2008); methane (CH_4) leakage (Aschmann *et al.* 2010); concurring interests for land and resources for food and feed (Jering *et al.* 2013); effects on ecosystem services (Holland *et al.* 2015; Milner *et al.* 2016); misleading technical potentials (Smith P. 2014); data variability; and model uncertainty (Malça & Freire 2010; Whitaker *et al.* 2010). Some of these aspects resulted in questioning the carbon neutrality assumption (Rabl *et al.* 2007; Wiloso *et al.* 2016). This postulation is the main reason for the preferability of biomass over fossil fuels, which posits, in short, that the direct emissions from biomass conversion can be neglected because the same amount of carbon dioxide (CO_2) had been fixed from the atmosphere shortly before by plant growth (4.1.5).

Policy development, strategy decisions, and technology choices all call for the quantification of possible mitigation contributions. This need resulted in calculation instructions that are reviewed with each new criticism from the scientific community. The general approach (4.1) is to balance the overall life cycle GHG emissions (mainly CO_2, nitrous oxide, methane) from bioenergy versus those from fossil energy. Several guidelines exist for this approach, for example in the RED directive (European Parliament and the Council 2009), or in carbon footprint methodologies (BSI 2011), and GHG protocol standards (WRI & WBCSD 2011)).

Still, varying or even contradictory mitigation contributions from bioenergy/biomass usage are calculated. Such uncertainty of –as well as existing confidence in– results remains difficult to communicate to the broad public (Collins & Nerlich 2015), and impedes the transformation to a sustainable society.

Relevance of Agriculture for Climate Change and its Mitigation

Nations are regularly reporting their GHG emissions according to international rules (Doha amendment (UNFCCC 2012), Kyoto Protocol (UNFCCC 1998)). The emissions are allocated to different source categories, one of them being the agricultural sector. Whereas CO_2 is mainly emitted in Germany by the energy sector (Gniffke 2016), CH_4 emissions arise from agriculture, energy, and waste management, and N_2O from agriculture, industry, and energy.

In total, the agricultural sector itself is held responsible for nearly seven percent of the German GHG emissions (Figure 1.2). Its contribution to CO_2 emissions is low because LUC emissions are reported in another source category. By contrast, it is the main contributor to N_2O emissions (80 %; agricultural soils) and a relevant contributor to CH_4 emissions (nearly 60 %; livestock husbandry, manure management).

Besides its role as a supplier of CC mitigation options via biomass generation, in turn, the agricultural sector itself contributes to climate change and needs to optimize its activities in order to provide biomass for downstream processes without emitting much additional GHGs.

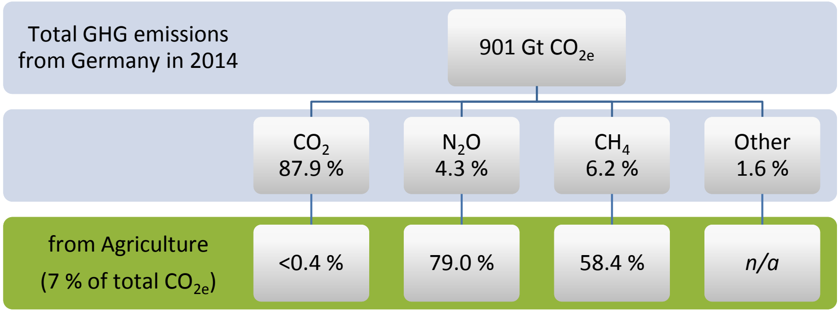

Figure 1.2: Share of GHG gases that are relevant in the source category 'agriculture' compared to the total GHG emissions (as CO_2 equivalents), and their fractions within this source category (without CO_2 form LULUCF; from German National Inventory Report (Federal Environment Agency 2016; Gniffke 2016))

Biomass Types from Agriculture for Transformation to Energy and Materials

Agriculture and its downstream industries provide a wide range of biomass types: from crops purposely grown for energy generation, for instance fast-growing wood (see 3.1.1), or maize (see 3.1.2); to crop residues (for instance cereal straw); or industry co-products as for example press cakes from oil processing; as well as organic waste, for instance from households or slaughtering. From the different types of biomass, in particular the woody one is often seen as promising source of bioenergy that could deliver large quantities at an overall positive climate impact (European Commission 2014). Biomass for material usage comprises oil, starch and sugar, and medical as well as fiber plants (3.1.3). However, acreage for bioenergy (see 3.2) is eight times that for materials usage (FNR 2016a).

In short, human society has recognized the need to change to a sustainable economy and has implemented first steps, for example by focusing on biomass usage. However, in doing so, it needs to account for the complex relations between biomass production and biomass usage. Implemented strategies need to be continuously monitored with adequate methodologies.

2 Research Objectives and Structure of the Thesis

Against that background, this publication-based dissertation aims to contribute to the discussion on reliability of CC mitigation assessment of biomass application in an increasingly bio-based, low-carbon economy that also fulfils sustainability constraints of resource conservation. It aims to answer the following questions:

- How much CC mitigation can we expect from bioenergy and how reliable are such –if existing– mitigation potentials?
- What are appropriate baselines against which GHG fluxes are balanced in the agricultural context? Is there a need to include more processes into the assessment?
- How relevant are uncertainties on the bioenergy process level if biomass usage is assessed in a broader context as in bioeconomy or in multi-criteria assessments?
- Do alternatives to mitigation analyses exist in order to address agriculture-specific characteristics of biomass generation and transformation?

The research questions are addressed via three separate case studies (see articles in sections 6.1-6.3), that integrate into the dissertation structure and relate to specific questions (Figure 2.1).

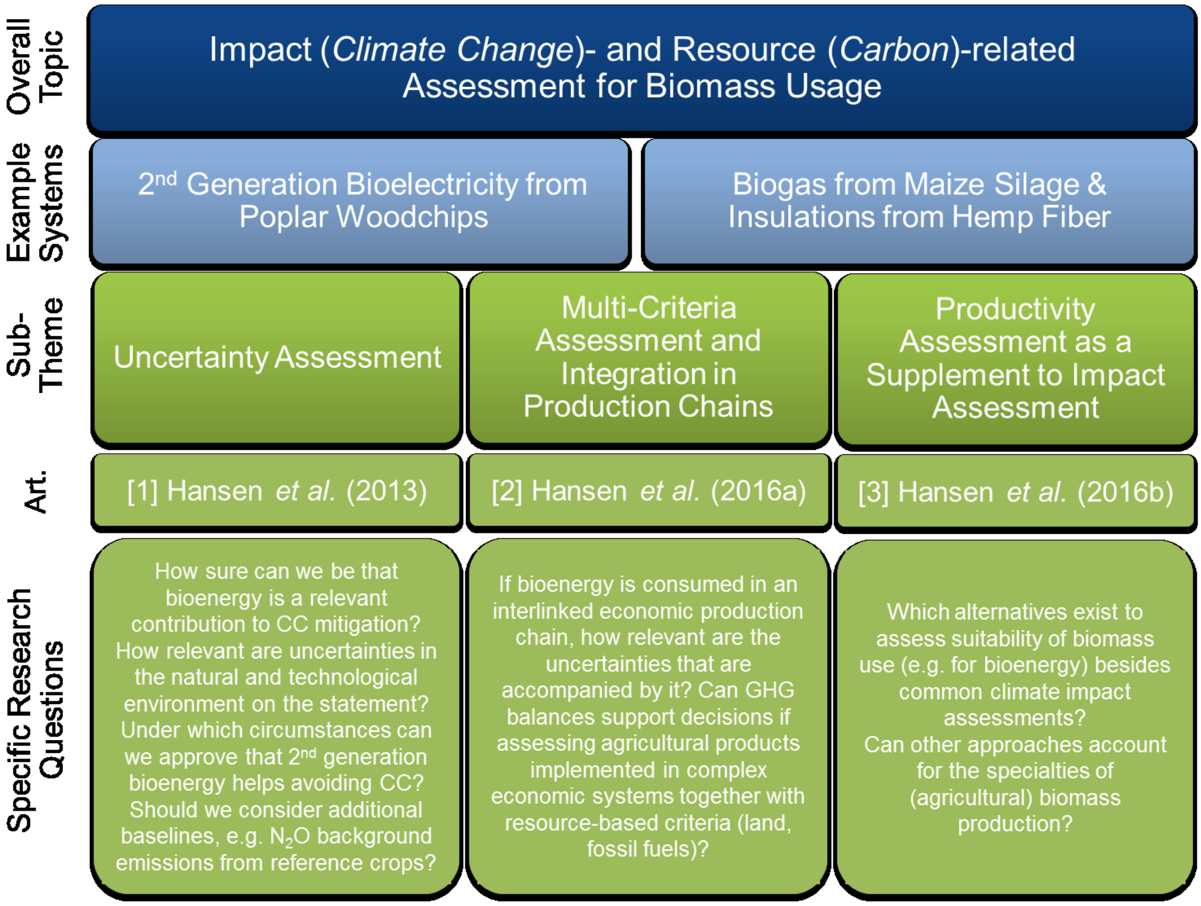

Figure 2.1: Integration of example systems in the case studies and sub-themes into the dissertation structure, and specific research questions

The dissertation is based on the following three articles (6.1- 6.3):

[1.] Hansen, A., A. Meyer-Aurich and A. Prochnow (2013) "Greenhouse Gas Mitigation Potential of a Second Generation Energy Production System from Short Rotation Poplar in Eastern Germany and its Accompanied Uncertainties" *Biomass Bioenergy* 56: 104-115 DOI: 10.1016/j.biombioe.2013.05.004

[2.] Hansen, A., J. Budde and A. Prochnow (2016a) "Resource Usage Strategies and Trade-Offs between Cropland Demand, Fossil Fuel Consumption, and Greenhouse Gas Emissions—Building Insulation as an Example" *Sustainability* 8: 613 DOI: 10.3390/su8070613

[3.] Hansen, A., J. Budde, Y. N. Karatay and A. Prochnow (2016b) "CUDe - Carbon Utilization Degree as an indicator for sustainable biomass use" *Sustainability* 8: 1028 DOI: 10.3390/su8101028

Article 1 (Hansen *et al.* 2013)(Uncertainty of Climate Impact from Second-Generation Bioelectricity) analyses the GHG emissions from a second-generation bioenergy generation pathway from gasified wood chips which are used in a combined heat and power plant (CHP) for electricity and heat generation. It estimates the pathway's potential to contribute to climate change mitigation in comparison to the recent electricity-mix. This is exemplarily done for a case study with a regional focus on Eastern Germany. The GHG mitigation analysis is performed for all management processes necessary to grow poplar as short rotation coppice, using a material flow analysis model and long-term emission data from a short rotation coppice (SRC) plantation of poplar. It addresses methodological aspects as the appropriate baseline choices for land use change (Soimakallio *et al.* 2015; Brander 2016) for such analyses by exemplarily including also N_2O emission data from a reference crop. It uses a Monte Carlo (MC) analysis to identify uncertainty and variability effects on the overall mitigation result. From that analysis, it specifies possible ranges how climate friendly such (second) energy generation from agricultural wood could be.

Acknowledging that energy is not an end in itself but instead is always embedded in bigger economic chains as a production factor, article 2 (Hansen *et al.* 2016a)(6.2) builds on the detailed process knowledge from article 1 and extends the view by incorporating the energy generation unit process exemplarily into two complex production systems. The production systems in this second case study have been defined in order to answer the question how agricultural land as well as fossil fuels –both being limited resources (see Figure 1.1)– should be used most efficiently, with the additional requirement that they have a low climate change impact. Each system exemplifies the identical promising way to mitigate climate change by providing building insulations: one grows material on the cropland (fiber hemp) and uses the fossils fuels for production energy, the other grows energy crops for production energy (SRC or biogas from maize) and uses the fossil fuels as a feedstock for material (polystyrene). This study has a wider geographical scale because co-products are traded Europe-wide.

Finally, considering the methodological difficulties faced in these two case studies, and to promote the understanding of agricultural production being essential –besides than just climate change–, article 3 (Hansen *et al.* 2016a)(6.3) presents an assessment approach of biomass usage by switching from the impact-oriented approach 'GHG mitigation' to a productivity/resource-oriented approach.

The following chapters 3 and 4 (Biomass from Agriculture and State of the Art) summarize basic agricultural and methodological background, and provide additional information that has not been included in depth into the articles. These sections could be skipped by readers already familiar with mitigation assessments and uncertainty calculations.

Chapter 5 briefly summarizes the approaches chosen in the articles, which are then presented in chapter 6. Chapter 7 jointly discusses the articles regarding the research questions, followed by Conclusions and Outlook (chapter 8). Chapter 9 lists all references cited in the document, some doubling those already listed in the articles' references sections. Supplement 12 presents information from a literature review on sequestration reports under bioenergy crop plantations.

3 Biomass from Agriculture

3.1 Energy Crops and Products

3.1.1 Woody Biomass Grown on Agricultural Sites - Short Rotation Coppice

From the different biomasses, especially the woody one is often seen as promising source of bioenergy that could deliver enough amounts jointly with an overall positive climate impact (European Commission 2014). Several tree species can be managed on agricultural sites in temperate climate using different cutting cycles (short rotation coppice management). Cutting cycles can vary between 3-10 years, depending on available harvest technology and intended way of use (Dimitriou & Rutz 2015). The trees can either be harvested and chipped in one step or separately as rods and chipped later (KTBL 2012). Poplar (*Populus spec.*), willow (*Salix* spec.), and black locust (*Robinia pseudoacacia* L.) are the common species in a European context whereas *Eucalyptus* spec. is of interest also in the Oceanic region (Sims *et al.* 2001). Such fast-growing trees grown on agricultural sites are considered as perennial crops, so no LUC to forest occurs through the establishment of a plantation. Poplar is characterized by its fertilization regime, it needs hardly any nitrogen (N) fertilizer, and its yields might even react negatively to it whereas willow yields react positively (KTBL 2012).

Carbon stocks in soils under SRC are often assumed to increase. However, experimental evidence of soil carbon stock changes painted a blurred picture, as Don *et al.* (2012) demonstrated in an overview of sequestration rates for *Miscanthus* and SRC plantations (1622±1586 kg CO_{2e} $ha^{-1}yr^{-1}$ for SRC on previous cropland). SOC stocks were found to possibly initially decrease (Hansen 1993; Jug *et al.* 1999; Grogan & Matthews 2002; Arevalo *et al.* 2011), especially on freshly-planted sites. Such stock decrease implies that climate-effecting carbon compounds were emitted into the atmosphere. Similarly, Laganiere *et al.* (2010) found in their comprehensive literature review of woodland afforestation that SOC levels after long-term re-establishment of trees may increase, decrease as well as stay invariable. They pointed out that SOC stocks appear to decline during the first years after planting. This could be due to the initial low biomass C input rates from young plantations or to the accelerated mineralization after site preparation (Poeplau *et al.* 2011). The reasons for such fuzzy soil C stock reports are manifold and have been discussed intensely. They range from a) discrepancies in the soil-depth increments that were probed (Schlesinger & Lichter 2001; Kravchenko & Robertson 2011; Powlson *et al.* 2011; Schmidt *et al.* 2011); b) the not yet equilibrated status of the soil carbon pool (Sanderman & Baldock 2010) ; c) different measurement methodologies (Poeplau *et al.* 2011); d) experimental design issues (Kravchenko & Robertson 2011); and e) time aspects (Cherubini *et al.* 2011; Garten Jr *et al.* 2011; Powlson *et al.* 2011) (please see supplement 12.1 for more details).

In Hansen *et al.* (2013), potential SOC increase was considered in the MC analysis by following Fritsche and Wiegmann (2008) who had reported possible stock increases of 27.5 t CO_{2e} ha^{-1}. This amount was annualized over sixteen years which was the assumed plantation standing time. Using a rectangular distribution R(-1719, 0) (kg CO_{2e} ha^{-1} yr^{-1}), the MC analysis showed the importance of SOC assumptions (nearly 80 % of the mitigation potential uncertainty derived from SOC uncertainty; Table 6.6). Accordingly, in (Hansen *et al.* 2016a), SOC effects were taking into account only in a scenario analysis of indirect effects. No direct SOC changes were considered due to high uncertainty of the long-term effect (Walter *et al.* 2015).

Many reasons exist why farmers would cultivate SRC (Kudlich 2011; Keutmann *et al.* 2016). Workload would be transferred to less work-intense times during winter, and after the initial plantation establishment only few maintenance measures would be necessary. Accordingly, sites distant to the farm could be cultivated economically. If long-term contracts with consumers are signed, a sure income could be generated, or else the harvested biomass could be used by the farmer directly.

Biodiversity in plants (Baum *et al.* 2012) and invertebrates (Rowe *et al.* 2011) in agricultural landscapes is assumed to be improved from such plantations, and also bird populations might be positively influenced (Fry 2011; Riffell *et al.* 2011). However, such effects may vary spatially (Eggers *et al.* 2009; Costanza *et al.* 2016). Even though such positive effects in rural areas are anticipated, in 2015, only 11,000 ha with perennial woody energy plants have been reported for Germany, compared to 6,000 ha in 2011 (FNR 2016a). This is inconsistent with the estimation of the Biomass Strategy Plan from 2010 (Kenkmann 2010) which assumed this acreage in the federal state of Brandenburg alone. Similar adoption problems have been reported for example from Scotland (Warren *et al.* 2016). Uncertainty in profitability has been mentioned as a reason for such hindrance in broad-scale deployment of SRC (Keutmann 2012; Lazarus *et al.* 2015).

Biomass yields increase over the plantation standing time and are usually expressed as mean annual increments, ranging from 8-36 t fresh matter (dry matter content 45 %) $ha^{-1}yr^{-1}$ for poplar wood chips depending on rotation length (KTBL 2012). Yields can either be assessed by destructive measurements (i.e. harvest) or non-destructive modelling approaches (Hauk *et al.* 2015).

SRC wood chips are used as feedstock in Hansen *et al.* (2013) and Hansen *et al.* (2016a) (6.1 and 6.2).

3.1.2 Annual Maize, for Example as Feedstock for Biomass Digestion

Maize (*Zea mays* L.) is still a debated annual energy crop in Germany. It has originally been and still is an important crop in South America and Africa for human nutrition (Bonavia 2013; FAO 2016). In Germany, in 2015, approximately two thirds of the maize cultivation area have been grown for cattle feed (as maize silage) and as kernels for pig and poultry, whereas one third of the area has been cultivated for maize silage for biogas generation. Since 2006, the total maize area has increased by approx. 750,000 ha due to energy maize cultivation (FNR 2016b). Especially an increase of maize acreage in previously grassland-dominated regions has been criticized. Converting grassland to maize acreage might result in CO_2 emissions from SOC changes (Fritsche & Wiegmann 2008) as well as might have other impacts as for example on bird biodiversity (Blank *et al.* 2016).

Maize does not tolerate low temperatures and hence is sown in spring, calling for winter catch crops or other management approaches to avoid soil erosion (Vogel *et al.* 2016). Maize for biogas production is harvested in summer when plants are still green, and conserved through ensiling (3.2).

Maize is used as biogas feedstock in Hansen *et al.* (2016a) and Hansen *et al.* (2016b) (6.2 and 6.3).

3.1.3 Natural Fibers from Hemp, for Example as Raw Material for Building Insulation

Besides for food, feed and bioenergy, agricultural crops are increasingly grown for industrial use, totaling approximately 270,000 ha in 2015 in Germany. Of this area, nearly 750 ha are cropped with fiber plants like hemp (*Cannabis sativa* L.) or flax (*Linum usitatissimum* L.) FNR (2016a). Hemp had been grown in Europe for cloth and ropes at least since the 1500s and in China since 6,000 years (Amaducci *et al.* 2015). Its importance decreased with the upcoming of synthetic materials. After the phasing-out of subsidies for hemp processing, German hemp acreage decreased even further from 4,000 ha in 1999 to 424 ha in 2012 (Kulicke 2013). However, hemp has just recently been assigned a high potential for bioeconomy use even though it is still a niche crop (Amaducci *et al.* 2015). This is due to its characteristic of being a multi-output crop, as its fibers, seeds as well as shives[3] have a market value. Its fibers are used in technical textiles, for example in automotive composites (Flake *et al.* 2000) or in building insulation (Danner 2010). Depending on the target application of the hemp products, hemp cultivation as well as cultivars should be chosen to yield maximum economic output (Amaducci *et al.* 2015). In turn, regional market segments vary for hemp co-products and their possible substitutes (hemp seed for human nutrition in Canada or for bird feed in Europe; see discussion on co-products and their possible substitutes in 2.2.4 in Hansen *et al.* (2016a)).

Hemp insulations are often produced from hemp long fibers. These are bonded to mats, consisting of a mixture of hemp and polyester fibers, and an additional impregnation of sodium hydroxide as a flame retardant (Bos 2010). Climate impacts stem from energy generation for the production process as well as from provision of the additional ingredients.

[3] Woody core of the stem, consisting of lignified cells and woody fibers; also called hurds (Amaducci *et al.* 2015)

Land use impacts are esteemed moderate. Hemp can be grown on less productive soils and marginal areas. Its cultivation intensity is low (fertilization at levels of 50-100 kg N ha^{-1}, and little need of weed control (Amaducci *et al.* 2015)). The latter is the reason for hemp being reported as positive within crop rotations. Climate impacts are not to be expected from direct LUC, as above and below-ground biomass as well as SOC are not expected to change in comparison to other annual crops.

Hemp fibers are used as insulation material in Hansen *et al.* (2016a) and Hansen *et al.* (2016b) (6.2 and 6.3).

3.2 Energy Transformation Options for Biomass

The energy content of biomass can be made available via several pathways. These have been distinguished –especially in the context of biofuel generation, however not exclusively– regarding the type of biomass feedstock and the transformation technology (WBGU 2009). A common distinction is that between first-(1G), second-(2G) and third-generation (3G) bioenergy. Whereas the first uses protein-rich or fat-containing agricultural products like grains or seeds, 2G processes non-digestible, lignocellulosic or at least agricultural co-products or biomass waste, and 3G biofuels are those made from algae (Stephens *et al.* 2010; Chaudry *et al.* 2015; Jambo *et al.* 2016; Kumar *et al.* 2016) or hydrogen from biomass (Bauen *et al.* 2009). Especially the switch from 1G to 2G was done as a reaction to the discussion if edible biomass should be used for energy generation ('Food vs. fuel' (Rosillo-Calle & Hall 1987; Tomei & Helliwell 2016)). Besides ethical also sustainability issues were raised against 1G (Dauber J 2012), even though some of this criticism was found to apply similarly to 2G (Mohr & Raman 2013). Another often used distinction is based on the technology that is used for the biomass transformation (Table 3.1).

Table 3.1: Distinction between bioenergy types (Bauen *et al.* 2009)

Type	Technology	Products
1G	Fully developed	Bioethanol from sugar/starch plants, biodiesel from oil seeds and animal fat, biomethane from anaerobic digestion of wet biomass
2G	Bio-/thermo-chemical conversion pathways (at demonstration stage)	Biofuels (for example ethanol, butanol, syndiesel) from ligno-cellulosic biomass (fibrous biomass as straw, wood or grass)
3G	Early research & development stage	Biofuels from algae, hydrogen from biomass

1/2/3G – First/second/third generation

This thesis also follows an technology-based understanding as WBGU (2009), who define 2G as synthetic energy carriers that have been produced via thermo-chemical processes such as gasification or pyrolysis. In this sense, 2G energy in the case studies (6.1 and 6.2) is energy that was generated from lignocellulosic biomass from high-yielding, perennial energy crops, being specifically of the non-food and non-feed type.

The gasification process is seen as a very efficient option to make the energy from biomass available and its basic principles are meanwhile well understood (Puig-Arnavat *et al.* 2010). The basic technology is centuries old and was increasingly re-used during World War II. Numerous reactor types exist (Breault 2010). The resulting gaseous product can be used in several energy generating technologies as for instance gas turbines or can be further processed to other products (Rauch *et al.* 2014). A world-wide overview of recent gasification projects as well as fact sheets on gasification and biomass resources can be derived from an IEA database (IEA Bioenergy).

1G technology in this thesis (6.2 and 6.3) is the methanation of biomass. This technology relies on anaerobic digestion of biomass by bacteria which yields gaseous metabolites. Maize as feedstock is usually conserved by ensiling after its harvest. During ensiling, bacteria feed on the sugars and starch and the residual metabolites lactic and acetic acid reduce the pH value. The silage is transferred to a fermenter where microbial communities digest the biomass and produce CH_4 and CO_2. This biogas can be transformed to heat and electricity in an on-site power plant. Alternatively, the biogas can be upgraded to a higher CH_4 content and be fed into the local gas grid. During this processing, leakages may occur at several intermediate steps (see 6.3.3 in (Hansen *et al.* 2016b)).

4 State of the Art – Mitigation Calculation and Uncertainty, Sustainability & Productivity Assessment of Biomass Usage

4.1 Climate Change Mitigation Assessments of Biomass Usage Systems

4.1.1 LCA as Basic Approach

The common methodological basis for mitigation analyses is the LCA methodology. This impact-oriented assessment of products and services comprises of four main steps: I) Goal and Scope Definition, II) Inventory Analysis, III) Impact Assessment, IV) Interpretation (DIN EN ISO 2006b). The basic idea of LCA is that whole life cycles must to be considered for a meaningful assessment, starting from the resource extraction to final disposal. A system boundary terminates the processes that are included into the assessment and balanced regarding a functional unit (FU). LCA targets an broad assessment of different environmental impacts (global warming, stratospheric ozone depletion, acidification, eutrophication, etc.) in order to avoid the reduction of one impact at the expense of another.

4.1.2 Mitigation Assessment of Biomass Usage Systems

For mitigation analyses, only the 'global warming' impact is chosen out of this list. Accordingly, the inventory analysis step concentrates on the occurrence of those gases that contribute to global warming. Whereas the total list of known GHG comprises of nearly 90 different gases (IPCC 2011), the most relevant in the context of agriculture and of biomass usage/bioenergy, are

- Carbon dioxide (CO_2)
- Methane (CH_4)
- Nitrous oxide (N_2O)[4].

A GHG inventory of a bioenergy generation pathway summarizes the emissions from relevant agricultural activities (Figure 4.1), for example from cultivation (direct emissions from fuel and fertilizer use; direct land use change (see 4.1.5); indirect emissions from fertilizer, fuel or pesticide production; indirect land use change (see 4.1.5), from biomass transport, and from biomass conversion (emissions from use of production factors or production losses; CO_2 from biomass is usually neglected due to the neutrality assumption, see 4.1.5).

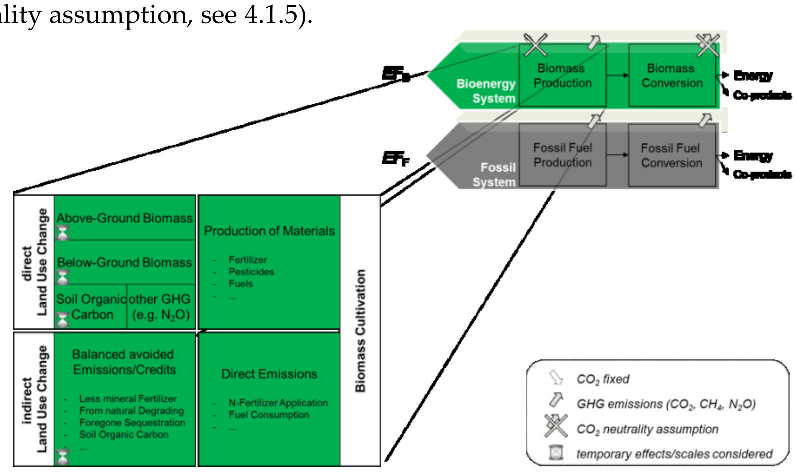

Figure 4.1: Greenhouse gas (GHG) mitigation analyses - overview of methodological approach and caveats (CO_2 neutrality, considered processes, temporary effects, etc.)

[4] According to the Kyoto Protocol (UNFCCC 2012), also hydro-/perfluorocarbons, NF_3 and SF_6 have to be reported for source categories in the National Inventory Reports.

Their amounts are aggregated to the single category indicator 'carbon dioxide equivalents' (CO_{2e}) by multiplication with their global warming potentials (GWP) (IPCC 2011). The GWP of a gas characterizes its radiative forcing relative to that of a reference gas (CO_2) for a specific time horizon. Usual time horizons are 20, 100 and 500 years, whereas the GWP_{100} is the most widely used even though no scientific argument exists for that (Myhre 2013). Due to an increase in scientific knowledge, the GWPs have been adjusted over the last two decades (Table 4.1).

Table 4.1: Modification of global warming potentials (GWPs) of CO_2, CH_4 and N_2O for time horizons of 20 and 100 years in the IPCC assessment reports (IPCC 2011; Myhre 2013)

Assessment Report (Year of publication)		SAR (1995)		AR4 (2007)		AR5 (2013)	
Time Horizon [years]		20	100	20	100	20[a]	100[a]
Greenhouse Gas							
	CO_2	1	1	1	1	1	1
	CH_4	56	21	72	25	84	28
	N_2O	280	310	289	298	264	265

SAR – Second Assessment Report; AR4/AR5 - Assessment Report 4/5
[a] no climate feedback included (see details in AR5 (Myhre 2013))

The balance result (=emission factor) EF_i relates the emissions over a complete energy generation chain (indices $_{F/B}$ – fossil/biogenic) to the generated energy. Common units are kg CO_{2e} MJ^{-1} or kg CO_{2e} kWh^{-1}. After the choice of the appropriate fossil reference system (4.1.3), GHG mitigation can then be either expressed as the mitigation factor MF_B as the difference between the fossil and biogenic emission factors EF_i

$$EF_F - EF_B = MF_B \tag{4.1}$$

or as relative mitigation potential MP_B [%]

$$\frac{EF_F - EF_B}{EF_F} \times 100 = MP_B \tag{4.2}$$

The subtractive nature of mitigation factors and potentials can mask the absolute height of climate impacts of the energy systems. Small (positive) ratings might result from (a) substituting a high-emitting fossil system by an also high emitting bioenergy system or (b) replacing an already low-emitting, efficient fossil system with a low-emitting bioenergy system (Table 4.2). Negative emission factors E_B in Table 4.2 can arise as the result of modeling choices: If for example sequestration effects from C stock increase are included in the analysis, they might overcompensate the emissions of the biomass processing. In such situations, resulting mitigation potentials can assign values greater 100 %.

Table 4.2: Masking effect of the subtractive nature of mitigation factors and mitigation potentials for decision support for technology choice, depending on height of emission factors

Mitigation Factor ǀ Mitigation Potential (MF_B) ǀ (MP_B)		Emission Factor (E_F)	
		High	Low
Emission Factor (E_B)	High	Low ǀ Small	Negative ǀ <0 %
	Low	Medium ǀ Medium	Low ǀ Small
	<0	High ǀ >100 %	Medium ǀ >100 %

E_F – Emission factor of fossil reference energy; E_B – Emission factors of bioenergy; $MF_B=E_F-E_B$; $MP_B=(E_F-E_B)/E_F$ [%]

4.1.3 Choice of Fossil Reference System

As the direct comparator in mitigation analyses, the choice of the fossil reference system has an important impact on results. From several existing options, either the use of the best available technique in terms of GHG emissions, that is from natural gas, has recommended, or otherwise the choice should clearly relate to the scope of the study (Cherubini 2010).

For Germany, a comprehensive model exists that maps the substitution options of different renewable energies within the existing power grid (Klobasa *et al.* 2009; Klobasa & Sensfuß 2016). It provides substitution shares [%] for these renewable energies in order to correctly represent which amount of which fossil reference energy is being substituted. These factors may change over the years (see Table 6.2), for example due to price development of CO_2 certificates at the European Energy Exchange (EEX 2016) or due to power plant shut-downs. Consequently, also the resulting mitigation factors MF_B vary: Since the 1990s, the overall MF of renewable electricity in Germany has decreased from 311 g CO_{2e} MJ_{el}^{-1} in 1990 to 173 g CO_{2e} MJ_{el}^{-1} in 2015 (BMWI 2016), one reason being that also the fossil emission factors EF_F have decreased. The variability in these reference technologies has also been discussed in scientific literature for U.S. conditions (Venkatesh 2012; Gurney *et al.* 2016).

4.1.4 Multi-Productivity

Important for a meaningful comparison of biomass usage systems as well as for system comparison in general is, that systems might yield multi-faceted benefits (co-products). This applies as well to agricultural production systems. Several approaches have been proposed to deal with this multi-functionality, for instance in LCA. DIN EN ISO (2006a) requests to first try and refine processes, then to expand systems, or finally to allocate burdens to co-products. Especially in agriculture, a process refinery may not be feasible as one cannot produce grains without straw, or milk without calves. On the other hand, system expansion might lead to undesirable large systems due to the variety of possible pathways for co-products, and hence, allocation issues in agricultural LCAs cannot always be avoided (Mackenzie *et al.* 2016). In allocation, if possible, partitioning should reflect physical relationships, else other possible proportions as for example economic values. Especially for agricultural products, recently "biophysical" allocation has been proposed, for example by relating the energy intake in feed to the energy output in the products for laying hens (FAO 2014) or in dairy farms (International Dairy Federation 2010), but has already been strongly debated (Mackenzie *et al.* 2016). Despite its methodological challenges, it is especially this multi-functionality of biomass production systems that offers GHG reduction potentials as well as taking pressure from land (Dornburg 2004).

4.1.5 Baselines

Neutrality Assumption of Bioenergy

GHG inventories do usually not account for direct CO_2 emissions from biomass conversion (Cherubini *et al.* 2009). Bioenergy or biofuel, respectively, receive a 'renewable' bonus in the way that both direct emissions during use as well as carbon fixation by biomass growth is neglected (Figure 4.1). This is termed *carbon neutrality*. In terms of CC mitigation, this might be debatable because on the short run, atmospheric carbon concentrations might increase, no matter if biomass or fossil fuel is burned (Cherubini *et al.* 2009; Sedjo 2011). In some bioenergy systems, the carbon might have been captured by the plants shortly before the combustion process and was hence called immediate carbon-neutral (McKechnie *et al.* 2011).

Challenging the general neutrality assumption began (e.g. (Rabl *et al.* 2007; Johnson 2009; Searchinger 2010) and continued over the years (EEA (SC) 2011; Don *et al.* 2012; Smith & Searchinger 2012), when it became obvious that biomass cultivation and transformation for bioenergy might induce climate-relevant emissions from its different sub-processes (for example CH_4 emission during ensiling and digestate storage (Herrmann *et al.* 2011; Murphy *et al.* 2011) or from wood chips piles during drying (see references cited in Whittaker *et al.* 2016). The neutrality approach has been criticized especially in the context of forest biomass (Haberl *et al.* 2012; Schulze *et al.* 2012).

Smith and Searchinger (2012) claimed that the postulation of C neutrality is adequate as long as 'additional carbon' is used, justifying the 'renewable bonus' (Searchinger *et al.* 2009; Searchinger 2010; Smith & Searchinger 2012). These authors defined additional carbon (i) as carbon from biomass from additional plant growth on previously unproductive land, (ii) as carbon from plant debris or other renewable residues (e.g. wood residues that would decompose and thus contribute to CO_2 emissions to the atmosphere anyway) or (iii) as avoided carbon if emissions are reduced through indirect effects (for example consumption reduction). The latter was considered unlikely in a first-generation biofuel context (Smith & Searchinger 2012), whereas it might be a real option under specific regional circumstances: if for example SRC are established on low-yielding, marginal agricultural sites and consequently no indirect effects are induced (Keutmann 2012; Keutmann & Grundmann 2014). Also a scale-dependency of C neutrality was discussed and the necessity of regional analyses proposed in order to reveal the short- and medium-term effects of bioenergy generation (Zanchi *et al.* 2012).

As a reaction, some studies included the fixed carbon in their input-side of the life cycle inventory (LCI). The amount of carbon was either deducted from the biomass yield per hectare and the respective C content (Carpentieri *et al.* 2005; Kern *et al.* 2010) or calculated via photosynthesis equations (Roedl 2010). The reasons given were that the LCI results might later on be used in another context (i.e. within value chains) where it might be important to know the appropriate 'carbon backpack' of the biomass. Secondly, that the carbon might be fixed for unknown time scales, for example if wood in house constructions is broken down at some time in the future. Thirdly, biomass fired plants with carbon capture and storage technology (CCS) would not get any bonus (Rabl *et al.* 2007). A frequently used LCI database also distinguishes between fossil and renewable CO_2 in its data sets (ecoinvent: Frischknecht *et al.* 2005). However, it still remains challenging to derive biogenic emissions as they not necessarily equal the CO_2 amount that has been fixed by plant growth. Some C might have been emitted as more effective GHG, for example CH_4 or in VOC (see later in 7.3).

Other approaches included the aspect into the impact assessment instead of the inventory phase. Cherubini *et al.* (2011) for example suggested accounting for biogenic carbon emissions with a rotation-period based GWP_{bio} index that describes the carbon re-fixation by plant re-growth. Similarly, Johnson and Tschudi (2012) put forward a biomass opportunity baseline that allocates the C, which is fixed by the re-growth of a forest which was harvested for energy, to the energy generated by that specific harvest.

Land Use Change Effects of Biomass Usage/Bioenergy

Carbon neutrality is closely linked to the topic of land use change (LUC), which was (one of) the main arguments raised against C neutrality (Searchinger *et al.* 2008). Land use (LU) –which is often not clearly separated from land cover (LC) (IPCC 2000) – has been defined as

- land cover (observed physical & biological cover of land surface, as vegetation or anthropogenic objects) (for example in CORINE/Coordination of information on the environment (EEA 1995); Figure 4.2 b)
- being "characterized by the arrangements, activities & inputs people undertake in a certain land cover type to produce, change or maintain it" (Di Gregorio & Jansen 2000)
- "The type of activity being carried out on a unit of land." (IPCC 2003)
- similar areas in terms of their socio-economic function (eurostat 2015)

WBGU (2000) suggested differentiating LU also regarding its intensity. IPCC (2006) adopted the six LU categories from the Good Practice Guidelines (IPCC 2003) for reporting on C stock changes and GHG emissions from LULUCF activities under the Kyoto Protocol (UNFCCC 1998), even though admitting that they are a mixture of LU and LC categories (Figure 4.2 (c)).

The terminology previously used for national GHG accounting changed from LULUCF (Land Use, Land Use Change & Forestry in IPCC 2003) to AFOLU (Agriculture, Forestry, and Other Land Use) in AR4. The most recent IPCC assessment report (AR5) integrated for the first time all land use into one chapter to consider all land-based mitigation options together, with an extra appendix on bioenergy (Smith P. 2014).

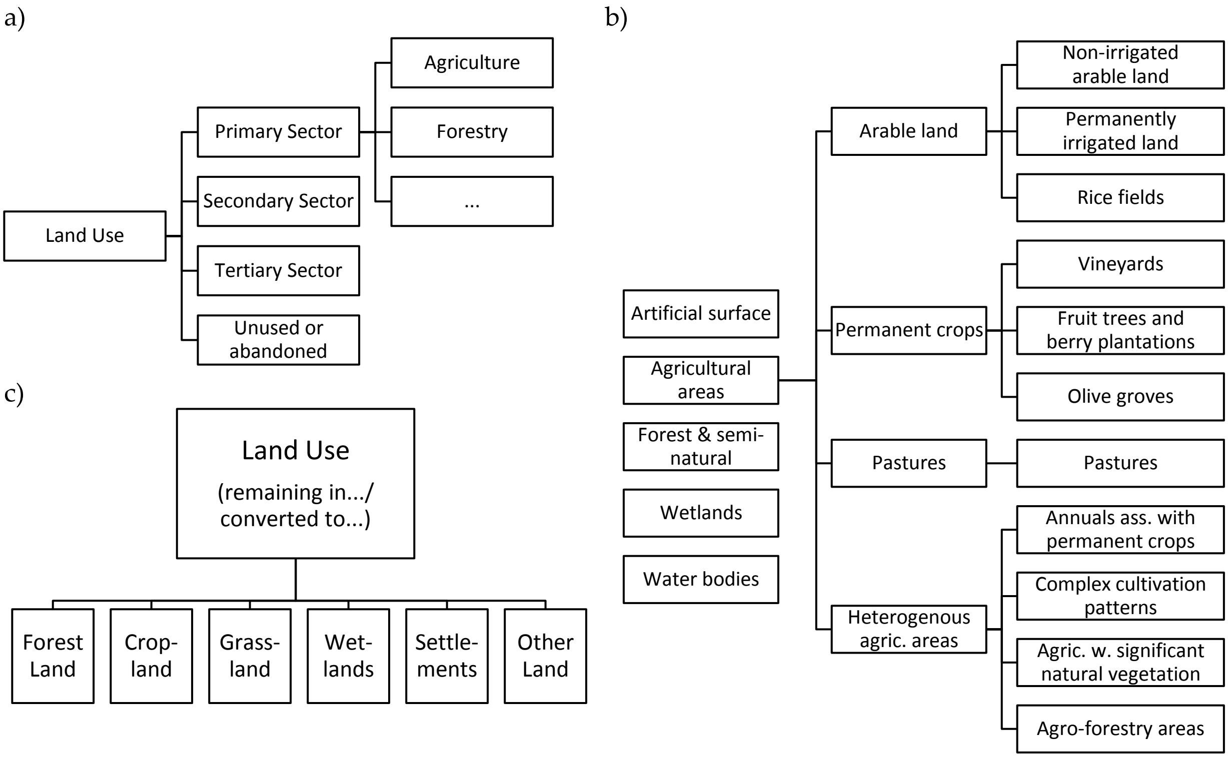

Figure 4.2: Variants of land use (LU) categorization: (a) LUCAS-Code (Land Use/Cover Area Frame Statistical Survey) (eurostat 2015), (b) CORINE Land Cover (CLC) nomenclature (EEA 1995), (c) LULUCF activities (IPCC 2003)

Generally, LUC denotes the occurrence of a transformation between one LU category and another which results in GHG emissions. The LUC might induce local changes in the affected plot in:

- Carbon (C) stocks
  - Above-Ground biomass (AGB)
  - Below-Ground biomass (BGB)
  - Dead Wood
  - Litter
  - Soil organic carbon contents (SOC)
- Land management
  - Intensification
  - Extensification

Stock changes can indicate that C fluxes to atmosphere took place, for example as CO_2 emissions. They can as well be a sign of C uptake from the atmosphere (sequestration) (see also later in 3.1). Changes in land management possibly result in N_2O emissions (from nitrogen fertilizer production & application) as well as in CO_2 emissions (from liming and fuel use). In the beginning of climate negotiations, also ordinary LU situations that resulted in emissions from stock changes were accounted for (IPCC 2000). Such emissions are described by the term 'direct LUC' (dLUC). At that time, the term 'indirect' solely denoted the indirect non-CO_2 emissions (i.e. N_2O) from deposition and leaching of N fertilization in agriculture.

Additionally to on-site –direct– effects, GHG emissions might be induced elsewhere via market effects. This is denoted today by the term 'indirect LUC' (iLUC). It is related to the situation where LUC is triggered in other regions and emissions result from that change (which are direct emissions there). The term was brought up by Searchinger *et al.* (2008) who claimed that bioenergy crop cultivation might result in conversion of forest land for cropland elsewhere, hence inducing stock changes and consequently emissions.

Whereas local CO_2 fluxes due to stock changes can be measured with static chambers (Clayton *et al.* 1994) or with Eddy-Covariance measurements (Lee *et al.* 2005), an adequate accounting for iLUC by models or other approaches has been considered difficult (Babcock 2009; Plevin 2010), impossible (Palmer 2011) as well as not helpful to avoid emissions (Finkbeiner 2014). Nevertheless, iLUC effects have been implemented in studies via land-based iLUC factors, for instance for UK food production (Audsley 2009), for bioenergy in general (Fritsche *et al.* 2010), for biomass for solid bioenergy from forests (Fritsche *et al.* 2014) or for agro-industrial residues for biorefineries (Tonini *et al.* 2016). Most of these approaches tried to reliably implement the topic into existing policies on bioenergy. Others do not distinguish between dLUC and iLUC (Nemecek *et al.* 2014).

In its arising in the context of bioenergy, the term iLUC has just been linked to climate impact. Quite soon its influence has also been acknowledged on biodiversity or on local land rights (RFA 2008; Fritsche 2013). Meanwhile, iLUC is considered not only for biofuels but for biomass in general (Schmidt *et al.* 2015) or for livestock farming (de Vries & de Boer 2010). iLUC effects on other assessment criteria, as for instance resource demand, have not yet been addressed in scientific literature as far as the author is aware.

Two accounting ways exist to quantify CO_2 emissions from LUC: land-based (i.e. account for stock changes in the accounting period in the land use categories) or activity-based (i.e. account for emissions/removals from respective activities during the accounting period) (IPCC 2000). Emissions other than CO_2 are usually calculated by activity-based methodologies.

A common understanding is that LUC happens between the LU categories cropland-grassland-forest, whereas a change in the cultivated crop (hence cropland remaining cropland) is not considered LUC, unless a relevant change in management practice (intensification/extensification) is associated to this crop change (IPCC 2000).

Within mitigation analyses, emissions from bioenergy generation are compared versus emissions from a fossil reference energy system. Any internal, agricultural baselines within the bioenergy generation chain have been rarely included into such analyses (Flake *et al.* 2000; St. Clair *et al.* 2008; Drewer *et al.* 2012). Flake *et al.* (2000) discussed the effect of different reference crops including set-aside land within the context of a biomaterial study. St. Clair *et al.* (2008) compared pre-harvest emissions from bioenergy crop cultivation (SRC, oil seed rape (*Brassica napus* L.), *Miscanthus*) to different land use baselines (conventional/reduced tilled winter wheat (*Triticum* L.), grassland, and broadleaved forest). As a result, they suggested growing bioenergy crops on previous cropland to fully exploit the GHG benefits. Clearly, their suggestion has not yet considered any iLUC effects. Tonini *et al.* (2016) included agricultural management baselines as indirect LUC effect into their iLUC factor to account for changes in emissions due to intensification.

An extensive literature review found no methodological agreement on how land use baselines should be incorporated into LCA studies for product systems involving land use (Soimakallio *et al.* 2015). The authors distinguish between four situations and suggest the use of different baselines (Table 4.3). They explicitly argue that a baseline is required for an impact-oriented assessment of physical flows and that case studies are required to study and evaluate implications of the different possible baseline choices.

Table 4.3: Baseline options for land use in LCA studies and their suitability (Soimakallio *et al.* 2015)

Baseline	Appropriate…
Zero	→ only if the land is at a natural or semi-natural steady state in the beginning of the life cycle studied
Business as usual (BAU)	→ only in cases where no further human interventions are included, such as abandoned agricultural land or natural ecosystems
Natural or quasi-natural steady state	→ where the accounting of emissions and removals starts from that steady state
Natural regeneration	→when the accounting does not start from a natural or quasi-natural steady state but from human-induced land use

'Change in soil organic carbon stocks' is one of those parameters which are responsible for the uncertainty of mitigation potential assessments when bio-energy crops are to be compared to fossil reference feedstock (Malça & Freire 2010; Brandão *et al.* 2011). So far, owing to the lack of long-term experimental data, analysts often use SOC change values which were derived for generalized land-use change (LUC) baselines (cropland, grassland, degraded land or forest; IPCC, 2006).

Taken together, GHG accounting of bioenergy and biomass usage should identify any relevant emissions in the complete energy generation pathway as indicated in Figure 4.1 and should furthermore consider existing uncertainties and variabilities. It is still common to assume carbon neutrality and –if applicable– to account for dLUC effects. Even though considerable progress has been made regarding the topic of indirect LUC effects and their modelling, they still remain quite uncertain and hence, are usually addressed within uncertainty assessments.

4.2 Uncertainty Assessment

4.2.1 What is Uncertainty?

Uncertainty as the opposite of certainty is generally understood as a situation where the outcome of the situation is not exactly known. In the context of natural sciences, such a situation might be represented by a system that has been described by a mathematical (computational) model with functions and parameters. Uncertain knowledge of the representation of the system outcome can arise from different sources.

Firstly, the system understanding might not be sufficient, that is, not all components of the situation might be known and hence not all (relevant) processes might have been integrated in the model including sufficient representation of scale and time effects. Also indirect effects like foregone sequestration and substitution effects on the market, as well as social or technological uncertainties (Hall *et al.* 2011) or the choice of a reference systems (e.g. fossil reference or agricultural reference crop) belong to this broad category. Secondly, not all (necessary) parameter values may be exactly known. Furthermore, parameters might be correlated, what requires uncertainty propagation methods. An example for the latter are variable biomass characteristics that may result in a variable outcome of a biomass conversion process (Allen *et al.* 2016).

Some distinguish between epistemic –that can be reduced– vs. aleatory –that cannot be avoided– uncertainty (Johnson *et al.* 2011). In that sense, a knowledge increase about a system allows including further aspects into the model, and hence could reduce its epistemic uncertainty. Another way would be to increase data certainty and improve parameter reliability. However, as one cannot eliminate uncertainty completely, methods were proposed to analyze and describe the amount of uncertainty in the model outcome.

In agricultural systems, natural and man-made systems interact. They are open systems that may vary in time and space. Hence, agricultural activities have to deal with numerous variabilities (climate, soil, yields, SOC trends, etc.) that could –detailed measurements provided– be described mathematically by probability density functions. Agricultural activities are as well uncertain due to human decisions (market-prices, stake-holder involvement, etc.). In the notation of uncertainty evaluation, variabilities are considered uncertainties.

Whitaker *et al.* (2010) distinguished in a review of 44 LCA studies on biofuels "three distinct sources of variation: (1) 'real' variability in parameters e.g. cultivation; (2) 'methodological' variability due to the implementation of the LCA method; and (3) 'uncertainty' due to parameters rarely included and poorly quantified.". This is nearly congruent with the conclusion of Malça and Freire (2010) who integrated (1) and (3) into *parameter uncertainties* that result from imprecise measurements, unrepresentative data and temporal and spatial variability, and distinguished on the other hand *scenario uncertainties* that result from normative choices in the modeling procedure (for instance choice of functional unit, or allocation method). Heijungs and Huijbregts (2004) gave a similar description. This work follows the understanding of the latter.

The difference between uncertainty and variability was considered important to communicate because it affects the reliability of scientific results in public and even among scientists (Lehmann & Rillig 2014). Also Hallegatte and Mach (2016) recently stressed that four aspects of uncertainty must be evaluated and communicated: probability ranges that can be narrowed with future research, unknowns that are linked to a deep lack of knowledge, uncertain reactions that depend on societal decisions and geopolitical events, and other areas of uncertainty that reflect random or chaotic features of the climate system.

For experiments in bioenergy research, first guidelines were published how uncertainty should be assessed (Casler *et al.* 2015). In representations of study results, uncertainty is often illustrated in numerical values as mean±standard deviation (SD), mean±standard error (SE) or as ranges. Sometimes, median values are presented. The inherent understanding of significant digits that already represent ranges –for example 100 indicating 95-105 (two significant digits) or 100 indicating 99. 5-100.4 (three significant digits) (Johnson *et al.* 2011)– is seldom stated. In graphs, uncertainty can be displayed by error bars, confidence intervals or box plots. In study summaries, often ranges are displayed.

4.2.2 Methods to Deal with Uncertainty in LCA

Starting in 1996, several quantitative approaches to deal with uncertainty have been applied in LCA, as inspected by Lloyd and Ries (2007). Methods included stochastic modeling, scenario modeling, fuzzy-data sets, interval calculations, Bayesian statistics as well as analytical uncertainty propagation. The most frequently-used stochastic modelling approach was Monte Carlo analysis, sometimes (combined with) Fuzzy-methodology. Scenario analyses were usually combined with other uncertainty analyses.

The situation is quite similar today, as a compilation of recent LCA studies in the context of agriculture and bioenergy shows (Table 4.4).

Table 4.4: Implementation of uncertainty assessment in studies on emission and mitigation of GHG from biomass usage (incomprehensive, chronologically ordered list)

Topic	Addressed uncertainties	Methods	Type	Ref.
GHG from bioenergy systems for emission trading	Emission factor, activity data	Error propagation	Case study	Ney and Schnoor (2002)
GHG & mitigation costs from dairy farms	GHG emission factors, enteric fermentation, cost and effectiveness of propionate precursors	Monte Carlo; triangular distributions; 1000 samples	Model-based study	Gibbons *et al.* (2006)
Avoided GHG when using different kinds of wood energy	Transport, number of GHGs included, technology, ±10 % change in each assumption at a time.	Sensitivity analysis	Case study	Petersen Raymer (2006)
Net GHG emissions of three firewood production systems in Australia	Growth rates, logging frequency, product recovery, efficiencies, distances, and others	Minimum-Maximum range	Case study	Paul *et al.* (2006)
Reliable ranking of scenarios from LCIA results (electricity from coal)	Confidence indices of a set of LCIA results	Modified fuzzy approach	Method and application example	Benetto *et al.* (2008)
1st generation biofuel (rapeseed oil)	Parameter (yields, fertilizer application rates, etc., SOC, GWP), Scenario (co-product allocation)	Monte Carlo; lognormal, Weibull, normal distributions; 10000 samples	Case study	Malça and Freire (2010)
New biomass conversion technologies for fuel, heat and power production compared to heat production in Austria from woody pellets	Prices, investment cost, efficiency, other	Monte Carlo; normal distributions; 1000 samples	Case study	Schmidt *et al.* (2010)
Economics of 2nd generation biofuels	Share of heat sales; for MC (efficiencies, prices)	Scenario analyses; Monte Carlo; triangular distribution; 5000 samples	Case study	Voets *et al.* (2011)
Policy development; biofuels	Inclusion of ILUC, share of biofuel, carbon tax	Monte Carlo; normal, uniform, lognormal distributions; 5000 samples; sensitivity analysis, Spearman rank analysis	Case study	Rajagopal and Plevin (2013)
GHG mitigation from 2nd generation bioelectricity from poplar wood chips	N_2O emissions from poplar and reference crop, allocation, transport, efficiency, yield, SOC, fossil reference, GWPs	Monte Carlo; several distributions; 5000 samples; Spearman rank analysis	Case study	Hansen *et al.* (2013)
Three alternatives of environmental, technological and policy factors on the resource efficiency of EU bioenergy production	Minimum GHG emission target, consideration of ILUC, technology and feedstock constraints, land constraints	Scenario analyses	Report	EEA (2013)
Feedstock logistic effects on GHG emissions from corn stover for bioethanol	Yield, collection and storage, feedstock and commodity transport, preprocessing	Monte Carlo; lognormal distribution; 1000 samples	Case study	Nguyen *et al.* (2014)

Table 4.4 –continued–

Topic	Addressed uncertainties	Methods	Type	Ref.
Management variants for willow chips	Yield, belowground carbon sequestration, litterfall and leaf nitrogen content	Monte Carlo; normal distributions;	Case study	Caputo *et al.* (2014)
Policy design for advanced biofuels	GHG emissions, land and water use, biofuel production	LP model with Fuzzy constraints	Case study	Ziolkowska (2014)
Profit maximization from biofuel supply chains	Demand, price of end-products	Stochastic linear programming model; sensitivity analysis	Method	Azadeh *et al.* (2014)
Theoretical and technical biomass energy potential in Columbia	Availability of different biomasses	Monte Carlo; probability function depending on available data; 50000 trials, Latin Hypercube sampling using 1000 bins	Method and Case study	Gonzalez-Salazar *et al.* (2014)
Identification of optimal planting sites and rotation cycles for SRC poplar	Previous land cover, productivity, land costs, and genotype	Analyses of Variance for process-based model	Case study	Lazarus *et al.* (2015)
Identification of most significant factors for GHG reduction by electricity generation from wood pellets from forest residues	Change of drying fuel, internal fuel use, GHG emissions from storage, dry matter losses during processing, allocation	Minimum-Maximum range	Case study	Röder *et al.* (2015)
Land Use options in the US Great plains 1870-2000	Soil, livestock, tractor fuel, irrigation pumping, and fertilizer production; absolute and relative uncertainty	Error propagation	Case study	Parton *et al.* (2015)
Annual profit of a forest biomass power plant	Biomass quality, availability and cost, electricity prices	Monte Carlo combined with optimization model; scenario analysis;	Case study	Shabani and Sowlati (2015)
Comparison of different biomass-based electricity generation pathways	*n.a.*	Monte Carlo (details *n.a.*)	Case study	Xu *et al.* (2016)
Design of hybrid energy systems	Availability of renewable resources	Method of moments	Case study	Abdullah *et al.* (2015)
Comparison of GHG from U.S. production of three biobased polymer families	Fossil polymer, LUC, agricultural operations, milling, co-product treatment	Monte Carlo; normal, uniform, lognormal, and other distributions; scenario analysis; Spearman rank analysis	Case study	Posen *et al.* (2016)

ILUC – indirect Land Use Change; LCA – Life Cycle Assessment; LCI – Life Cycle Inventory; *n.a.* – not available; SOC – Soil organic carbon

General guidelines have been published (Williams *et al.* 2009; Johnson *et al.* 2011) that call for integration of uncertainty and sensitivity analysis results into the final presentation of studies on bioenergy (Cherubini *et al.* 2009). Sensitivity analyses are mandatory in Life Cycle Assessment (LCA) (DIN EN ISO 2006a; JRC 2010) and are also called for in the IPCC guidelines (IPCC 2006). Their aim is to identify which parameters are the most important ones whose uncertainty influences the results, especially if different systems are to be compared. In *sensitivity analyses*, parameter values are varied *ceteris paribus*[5] within a defined range, for example ±10 %, and the resulting range in the study results is calculated. From this, the most important parameters of the system can be identified, *i.e.* those whose uncertainty should be further reduced.

Monte Carlo (MC) analyses are another method to identify sensitivity hot spots and to assess overall uncertainty. They assign probability density functions to the different system parameters, draw a high number of possible parameter combinations and subsequently perform a high number of simulation runs. The results are then further analyzed regarding overall probability of the total system outcome. Its sensitivity to the uncertainty of the single parameters can also be assessed, for example by Spearman rank analysis. An important preliminary for using the approach is that parameters are independent and not correlated (Bojacá & Schrevens 2010). This is a pre-requisite that is difficult to safeguard, especially in very detailed models (Szyska 2009).

Especially for complex systems, the MC approach is often combined with *scenario modeling* to address model uncertainty. Scenario modeling can be a way to implement expert and stakeholder knowledge into reasonable model development (Bezlepkina *et al.* 2011). A variant of scenario analyses are *Minimum-Maximum-Analyses* (or interval calculations), which aim at the identification of the maximum range of possible results.

Other approaches that are not widely used are *analytical error propagation* as for example Taylor series expansions (Hong *et al.* 2010), *Fuzzy-data sets* or *Bayesian statistics*.

The usual approach in LCA is to perform ex-post analyses of uncertainty. However –especially in the context of decision making– an ex-ante assessment was proposed to improve communication between analysts and decision makers (Herrmann *et al.* 2014).

At present, several computer programs for LCA offer functionalities to deal with parameter variability and uncertainty of systems. Those functionalities allow performing uncertainty analyses, as for example MC-Analyses (Umberto (ifu&ifeu 1994-2011) used in Hansen *et al.* (2013), SimaPro (PRé Consultants 2008)), GaBi (PE International 2011) used in Saez de Bikuña *et al.* (2016) or Excel Add-ins (@RISK® (Palisade Inc. 2016) used in Meyer-Aurich *et al.* (2012) or Oracle Crystal Ball (Oracle Corp. 2016)) used in Gonzalez-Salazar *et al.* (2014). Data bases as for example ecoinvent (Frischknecht *et al.* 2005) apply approaches to transform qualitative information of flow information into quantitative ones, for example with a Pedigree matrix approach (Ciroth *et al.* 2016). Lettens *et al.* (2003) attributed a reliability score to describe the mean value for a unit process that has been derived from different data sources.

According to the state of the art, this dissertation applied a MC analysis to a bioenergy generation chain in Hansen *et al.* (2013) (6.1), whereas a scenario approach was chosen for the complex systems in Hansen *et al.* (2016a) (6.2). Article 1 included also differences in the agricultural reference crop baseline in the assessment (6.1.2).

As a fundamentally land-dependent activity, agriculture should be assessed not only with regard to its climate impact but at least also to its resource demand of land. This goes hand-in-hand with the methodological approach of mitigation analyses which calculate some of the climate impacts from land-based information anyway (fertilizer application per hectare for example). Another relevant resource demand is that of fossil resources, especially if comparisons to fossil-based systems are intended in the context of climate change mitigation. In Hansen *et al.* (2016a) (6.2), the SRC bioenergy pathway of Hansen *et al.* (2013) was implemented into a wider research agenda, in which these two additional indicators out of the comprehensive list of available indicators in sustainability assessments were considered.

[5] "other things held constant"

4.3 Assessment of Sustainability of Agricultural Products and Systems, with a Focus on Climate Impact, Land Use and Fossil Fuel Demand

GHG emissions are a standard component in the indicator sets of sustainability assessments and certification systems for agriculture (Table 4.5), independent whether on the product level, farm scale, supply chain or landscape scale (Hansen & Wolf 2015). GHG are assessed on the impact-level of assessments that is the inventory results are further aggregated to a model-based indicator (CO_{2e})(4.1.2), whereas land use and fossil resource demand are usually assessed at the inventory stage level (Hansen *et al.* 2016a). In society, the term 'sustainable land use' is often used synonymously to 'sustainable, organic or diverse agriculture', to 'good agricultural practice', or that no LUC takes place.

Table 4.5: Assessment methods for agricultural sustainability that include GHG emissions, land and fossil resource-associated indicators (digest as of 04/2015)

Acronym	Name	Indicators			Ref.
		Climate Change Mitigation	Land Use	Fossil Resources	
AgBalance©	BASF AgBalance Methodology v1.0	CO_{2e} CB^{-1}	ha (cropland; total land use)	kg Silver Equivalents CB^{-1}	Schoeneboom *et al.* (2012)
DLG	*DLG-Nachhaltigkeitsstandard*	GHG emissions GJ^{-1} or ha^{-1}		Qualitative criterion	DLG (2010)
EF	Ecological Footprint	*n/a* (indirectly by accounting for land necessary to absorb resulting CO_2)	gha (global ha)	*n/a* (indirectly by accounting for land necessary to absorb resulting CO_2)	Wackernagel and Beyers (2010)
INRO	INRO-Metastandard (*Initiative für nachhaltige Rohstoffbereitstellung*)	kg CO_{2e} kg^{-1}	Qualitative criteria	*n/a*	INRO (2013)
ISCC PLUS	International Sustainability and Carbon Certification	g CO_{2e} MJ^{-1}, g CO_{2e} t^{-1}, g CO_{2e} m^{-3}, or g CO_{2e} l^{-1} final product	Qualitative criteria	*n/a*	ISCC (2012)
KSNL/KUL	Criteria System Sustainable Agriculture	kg $CO_{2e}(ha{*}a)^{-1}$ or kg CO_{2e} GJ^{-1} Product	ha (median field size); percentage of organic fields	*n/a* (energy contents of all farm inputs aggregated)	Breitschuh *et al.* (2008)
LCA	Life Cycle Assessment	kg CO_{2e} FU^{-1}	m^2 FU^{-1} (land transformation), m^2a FU^{-1} (land occupation)	kg crude oil equivalents FU^{-1}, MJ FU^{-1}	DIN EN ISO (2006a)
FIPS	Area (*Flächen*) Input per Service Unit	*n/a*	m^2 FU^{-1}	*n/a*	Schmidt-Bleek (1994)
REDCert	REDCert	kg CO_{2e} MJ^{-1}	Qualitative criteria; location, e.g. geo-coordinates	*n/a*	REDcert (2010)
RSPO	Round Table on Sustainable Palm Oil	t CO_{2e} t^{-1} product	Qualitative criteria	Qualitative criteria	RSPO (2013)
SAFA	Sustainable Assessment of Food and Agriculture Systems	t CO_{2e}; qualitative criteria	percentage of land; qualitative criteria	Oil equiv. $Capita^{-1}$ (aggr. with other energy sources)	SAFA Initiative (2013)
SMART	Sustainability Monitoring and Assessment RouTine	Qualitative criteria (derived from SAFA)	Qualitative criteria (derived from SAFA)	Qualitative criteria (derived from SAFA)	FiBL (2014)
UGR	*Umweltökonomische Gesamtrechnungen/ Materialkonto*	$CO_{2(e)}$ (temperature corrected); CH_4 and N_2O per county	Percentage of organic area	t abiotic, used materials; t raw material equiv.	DESTATIS (2012)
USAC	Unilever Sustainable Agriculture Code	Qualitative criteria	Reduced land demand (ha); protected/ improved habitat area (ha FU^{-1}); improved soil quality on area (ha FU^{-1})	Qualitative criteria	Smith (2015)

CB – Customer Benefit; FU – functional unit; GHG – Greenhouse Gas; GJ – Gigajoule; *n/a* – not available; MJ – Megajoule

Table 4.5 provides an overview of existing assessment systems. Some of them were originally defined explicitly for the analysis of biomass for energy usage (ISCC (ISCC 2012), REDCert (REDcert 2010), RSPO (RSPO 2013)) and hence, focus solely on GHG emissions, whereas others see GHG just as one indicator in a large basket of important aspects (for example in KSNL; Breitschuh *et al.* 2008).

Whereas in Hansen *et al.* (2016a) (6.2), three distinct indicators (GHG emission, fossil fuel demand as well as land demand) were used to identify the most beneficial strategy in order to efficient resource use, Hansen *et al.* (2016b)(6.3) aimed to develop an integrated indicator, based on a productivity assessment.

4.4 Specific Climate Impact and Productivity Metrics for Biomass Usage

In order to assess whole nations or economic sectors regarding their contribution to CC or CC mitigation goals, other approaches than LCA or those in Table 4.5 above have been developed. They were derived from the productivity concept that is a common approach in economics and relate GHG emissions to economic output. Such approaches can be applied to an economic sector or a whole nation, for example *C Productivity* or its reciprocal *C Intensity* (Table 4.6; published as Appendix file to Hansen *et al.* (2016b))(6.3).

Table 4.6: Overview of some productivity approaches dealing with carbon (reproduction of Appendix Table A1 in Hansen *et al.* (2016b); references in brackets are listed in section 6.3.6)

Name	NPP/NEP	Carbon Productivity	Carbon Intensity	C Balance	MACC	S&P/IFCI Carbon Efficient Index	CSF	Carbon efficiency	*CUDe*
Denominator	Unit of area and unit of time	Unit of emitted CO_2[a] per period per country	Unit of sales		Unit of mitigated CO_{2e}		Unit of C in fresh biochar	Total C present in reactants	Carbon fixed in harvestable biomass
Numerator	Unit of generated energy (or biomass)	Unit of the specific value of GDP in the same period	C emissions		Unit of cost of a technology		Unit of biochar C after 100 years	Amount of C in product × 100	*Productive* C
Unit	g C m^{-2} yr^{-1}	Currency kg^{-1} $CO_{2emitted}$	kg C emitted/unit of sales $	%	Currency t^{-1} $CO_{2e\ mitigated}$		Dimensionless or %	%	%
Baseline	Usually one year	Arbitrary period length, often one year			Marginal cost and projected emissions of reference technology		100 years	Not stated	Adjustable
Description	Rate at which energy is converted into biomass	Used in economics; reciprocal of carbon emission intensity per unit of GDP [b]; "Reflects economic benefits yielding from per unit of CO_2 emission" [c]			"A MAC curve is a graph that indicates the marginal cost (the cost of the last unit) of emission abatement for varying amounts of emission reduction." [a]		Remaining C in carbonized biomass (biochar) after labile and instable fractions are released		Ratio of productive C to initial C_{in} in the biomass
Target audience	Science	Policy			Policy		Science	Pharmaceutical industry	Policy
Methodology	1. Measure biomass production (for example: destructive measurements /NPP; flux measurements /NEP; models) 2. Convert biomass dry matter according to C contents	1. Define period 2. Look up GDP and fossil resource use for that period 3. Calculate CO_2 emissions from resource use via emission factors			1. Define baselines (emissions in target year; technology) 2. Identify and describe possible abatement technologies and their costs for the target year 3. Plot abatement potentials on x-axis and costs per ton on y-axis		1. Measure C content in fresh biochar 2. Identify labile (after a few weeks) and instable (e.g., via accelerated ageing methods) C shares in original biochar		Please refer to Figure 6.8

Table 4.6 –continued–

Name	NPP/NEP	Carbon Productivity	Carbon Intensity	C Balance	MACC	S&P/IFCI Carbon Efficient Index	CSF	Carbon efficiency	*CUDe*
Benefits	In combination with modelling approaches, shortcomings (see below) can be overcome	Comparison of different nations possible Different development stages visible			Comparison of different nations possible Illustrative tool to present mitigation options		Intuitive to understand Reflects C sequestration Percentages in different products in multi-product systems can be summed up	simplified formula takes into account the stoichiometry of reactants and products of interest to the pharmaceutical industry where the development of carbon skeletons is key to their work.	Paradigm change to carbon being an asset of the bio-economy instead of a threat Reflects the C use efficiency of the conversion process
Shortcom-ings	Representativeness for analyzed biomes and attributed area critical Accounting for land use change	Only fossil CO_2 (no other GHG included, for example, nitrous oxide, CH4) Inherent connection to economic cycle and growth paradigm			Static representation of costs for single years; no allocation of costs to ancillary benefits of GHG mitigation; lack of transparency; poor treatment of uncertainty, inter-temporal dynamics, interactions between sectors (see details in a)		No additional necessary C for conversion processes considered CSF uncertain due to the wide range of assumed residence times of C remaining in the biochar after application to soils (293–9259 years[b])		No complete GHG assessment (only CO_2 and CH_4 included) CUDe is not directly related to output quantity No energy-related C input considered
Ref.	[64]	[c6,b23,a65] [a] uses CO_2 equivalents as a basis.	Only sparse information given in [66]	[67]	[68,a69]	[70]	[b62,71,72]	Acc. To [73] developed at GlaxoSmithKline (GSK); no original source available	This manuscript

CUDe—Carbon Utilization Degree, CSF—Carbon Stability Factor, GDP—Gross Domestic Product, GHG—Greenhouse Gases, MACC—Marginal Abatement Cost Curves, NPP/NEP—Net Primary Productivity/Net Ecosystem Productivity, S&P/IFCI—Standard & Poor's International Finance Corporation Indexes;
Unless otherwise indicated by superscripts, information was taken from cited References in the last row.

5 Short Overview of Approaches applied in the Articles of the Thesis

A bioenergy pathways was chosen that was about to leave the pilot scale state and was being introduced economically in 2011 (IEA Bioenergy). Its biomass feedstock (SRC) had been esteemed promising (3.1): Large energy providers pursued its cultivation (for example Vattenvall Europe AG, Ehm 2011), and intensively approached farmers at that time to grow SRC, and to close supply contracts. Furthermore, long-term GHG data were available from poplar SRC sites as well as from neighbouring reference plots, cultivated with the region's common cash crop rye (Kern *et al.* 2010).

For each manuscript, a comprehensive literature research was conducted prior to final methodological decisions; details are given in the respective articles.

The bioenergy pathway was modeled using an LCA Tool (Umberto® 5.6) (ifu&ifeu 1994-2011) for which unit processes existed from previous work (Möhlmann *et al.* 2000; Hansen *et al.* 2001). The tool includes a function to analyse uncertainty in the material flow models by MC analyses (4.2.2). According to the state-of-the-art of LCA studies (Table 4.4), a MC analysis was performed (see parameters and probability distributions in 6.1.2). The fossil reference system was taken from Klobasa *et al.* (2009). Their model presents the substitution effects of biomass in the German electricity mix (4.1). In Hansen *et al.* (2016a) (6.2), the results of the Umberto® model were merged in spreadsheets (MS Professional Plus 2010) with additional data from unit processes to represent two strategies for the insulation of buildings. For these strategies, a scenario analysis was performed due to the complexity of systems and resulting problems in safeguarding the independence of parameters for a MC analysis. Additional data were taken from LCA data repositories, for example ecoinvent (Frischknecht *et al.* 2005), and GEMIS (Fritsche & et al. 2014)(see details in the articles chapters). Additionally, telephone interviews were conducted with stakeholders for missing information, for instance with energy crop consultants who advise farmers on SRC implementation, or with feed producers who rely on specific agricultural ingredients for their product formula. These interviews have been qualitative and limited, therefore they cannot be considered as significantly representative for the complete sector.

The analyses used a life-cycle-based approach (see 4.1) and included the relevant production processes of crop cultivation and its production factors (fertilizers, pesticides, fuels, seeds). Allocation was avoided by system expansion where possible. Otherwise, a mixed approach was followed as in Eady *et al.* (2012). Detailed information on system boundaries and functional units are available in the respective articles (Hansen *et al.* 2013; Hansen *et al.* 2016b; Hansen *et al.* 2016a) (6.1-6.3). For the CC impact assessment, the three GHG out of the complete IPCC list were assessed that are the most relevant in an agricultural context: N_2O, CH_4 and CO_2. Further GHG from the Kyoto list (UNFCCC 1998) were included in a pre-study of the uncertainty assessment but were found to be irrelevant and omitted in Hansen *et al.* (2013). An extended LUC assessment as a variation of the usual LUC balancing approach was performed in Hansen *et al.* (2013). On the one hand, absolute N_2O emissions from unfertilized[6] poplar SRC plots were considered in the balance. In additional step, these emissions where balanced against the N_2O that would have been emitted from the reference crop rye. Common LUC assessments just consider stock changes in above and below-ground biomass as well as in soil organic carbon as land use change effects (see 4.1.5).

In distinction to these impact-oriented assessments of biomass utilization, a five-plus-one step approach was developed in Hansen *et al.* (2016b)(6.3). Following a process chain assessment (Figure 6.8), it identifies *productive* C for each transformation process. *Productive* C in an anthropocentric view was defined as C that yields a useful output, that is it is (i) transformed into marketable products or provides useful services (e.g. insulation material, or energy generation (direct benefit)), or (ii) performs important ecological functions (e.g. improves soil fertility (indirect benefit)). The ratio of the total productive C to C that was originally available in the biomass was defined as Carbon Utilization Degree (*CUDe*). Generic data for example calculation of productive carbon share in biomass chains (biogas and hemp fiber insulation) were compiled in spreadsheets and visualized with *e!*Sankey® 3.2.

[6] No N-containing fertilizer, but phosphorus and potassium fertilizer

6 Results – Articles Section

6.1 Uncertainty of Climate Impact from Second-Generation Bioelectricity

Published as:
Hansen, A., A. Meyer-Aurich and A. Prochnow (2013): "*Greenhouse Gas Mitigation Potential of a Second Generation Energy Production System from Short Rotation Poplar in Eastern Germany and its Accompanied Uncertainties*". Biomass Bioenergy 56: 104-115. DOI: 10.1016/j.biombioe.2013.05.004

6.1.1 Introduction

Second generation bioenergy is often seen as a means of reaching ambitious carbon dioxide (CO_2) emission reduction targets in industrialized countries and as a way to simultaneously increase the efficiency of land use [1]. It is frequently defined as energy produced from lignocellulosic biomass from agricultural and forestry plant residues and high-yielding, mainly perennial energy crops [2,3], being specifically of the non-food and non-feed type [4]. The latter characteristic describes the difference between second and first generation starch and oil crops. Second generation conversion technologies, which are still under development, are typically biochemical and thermo-chemical pathways [5].

The CO_2 emission-reductions, which possibly can be reached via bioenergy pathways, are usually calculated using the concept of mitigation potentials (see for example Refs. [6-8]). In principle, all greenhouse gas (GHG) emissions, including CO_2, methane (CH_4) and nitrous oxide (N_2O) of the complete bioenergy production chain E_B, are counterbalanced against those of a reference system based on energy from fossil and nuclear feedstock E_F. Typically, the emissions from combustion of the biomass are not accounted for, since they are captured by the plants shortly before the combustion process [1].

The mitigation effect E_F -E_B is then expressed as mass (kg carbon dioxide equivalents; CO_{2e}) divided by energy (here: Megajoule electricity generated; MJ). Financial incentives for renewable energies may depend on these mitigation calculations. In Germany for example, the act on granting priority to renewable energy sources (RESA) [9] requires minimum mitigation effects as pre-requisite for its guarantied feed-in tariffs for bioenergy. The conditions are defined in two ordinances [10,11] which implement the sustainability criteria formulated in the European Renewable Energy Directive (RED) [7]. Annex V of RED presents rules for the calculation of the GHG impact of biofuels, bioliquids and their fossil fuel comparators. These European and national regulatory frameworks aim at liquid biomass at the moment as does a draft standard [12]. The European Commission (EC) [13] recommended that member states use the RED sustainability scheme for solid and gaseous biomass accordingly, but proposed no binding methodology for calculating the GHG performance of solid and gaseous biomass. It just explicitly stated to use the life cycle assessment (LCA) method as described in the RED with some extensions. RESA was recently modified and now requires the ordinances to be adapted to include also solid biomass.

Existing approaches to assess the carbon (or GHG, respectively) footprints of products and services [14-17] agree on essential components of GHG inventories. Parameter and system uncertainties and variability are considered important, data quality should be stated, and relevance of values for the outcome of the analysis should be assessed [16]. Standard values for unit processes of supplying and processing biogenic feedstock, in case that no specific data are available, are provided by RED. These standard data are supposed to be conservative assumptions. Nevertheless, the underlying variability of parameters and assumptions is not transparent in published mitigation values. Albeit such frameworks exist, LCA's underlying methodological challenges make the comparison of different studies difficult [18,19]. Among the reasons for that are, for example, the choice of how the co-products are accounted for [20], the choice of the functional unit and the setting of system boundaries [21], the direct and indirect land use change (LUC) effects, the uncertainties regarding soil organic carbon (SOC) development, the choice of the conventional reference system, and the uncertainties in those fossil reference systems themselves [22,23].

Several ways for dealing with uncertainty in LCA have been suggested: (i) the scientific way (=do more research), (ii) the social way (=discuss and find consensus) and (iii) the statistical way, which means using methods from statistical theory and thus incorporating the uncertainty into the analysis [24].

Statistical methods are, for example, Monte Carlo (MC) simulations. These were recently used for the calculation of GHG emissions of the first generation biofuel rapeseed-oil, which displaced fossil diesel [25], and for the comparison of cash flows of two second generation technologies [26].

The majority of the existing studies focus on biomass as feedstock for second generation biofuels in mobile applications [2,5,27-31] and use generic data for the biomass provision. In contrast, we concentrate in this study on the conversion of ligneous biomass in a stationary electricity generation process and use data from an experimental poplar plantation site (*Populus maximowiczii* × *Populus nigra*) as a case study.

The objective is to determine the GHG mitigation potential of a second generation bio-electricity production system from poplar wood chips under German conditions, including reliable ranges, due to the uncertainties of the parameters and the underlying assumptions. The most influential of them, with regard to the overall uncertainty, shall be detected using MC simulations.

6.1.2 Methodology

Greenhouse gas mitigation of bio-electricity from short rotation poplar

The GHG mitigation effect of bio-electricity from poplar short rotation coppice (SRC) is expressed as the overall mitigation factor MF_B (kg CO_{2e} MJ^{-1}) with

$$MF_B = E_F - E_B \tag{1.1}$$

where E_F is the cumulative CO_{2e} emission MJ^{-1} from electricity generation via the country-specific fossil feedstock mix and E_B is the cumulative GHG emission when electricity is generated from poplar wood chips via gasification. E_B was assessed according to the valid national framework [10] and the suggested approach which was presented in Ref. [13]. A modification was made in the way that also N_2O emission savings from changed land management were considered. In the cited frameworks, LUC emissions or savings, respectively, so far only comprise those due to carbon stock changes. Generally, E_B includes GHG emissions from pre-chains, from farming operations of agricultural SRC cultivation and from the transport of the harvested biomass to the conversion site. The processes considered for the respective production chains are presented in Figure 6.1. More detailed assumptions and system descriptions for the production chains are given in the following paragraphs. For all processes the absolute amounts of the emitted greenhouse gases were converted to carbon dioxide equivalents with the actual global warming potentials (GWPs) for a 100-yr time horizon (GWP N_2O = 298; GWP CH_4 = 25 [32]). The functional unit is 1 MJ electric energy generated either via fluidised bed gasification of poplar wood chips or with the reference electric energy supply systems in Germany.

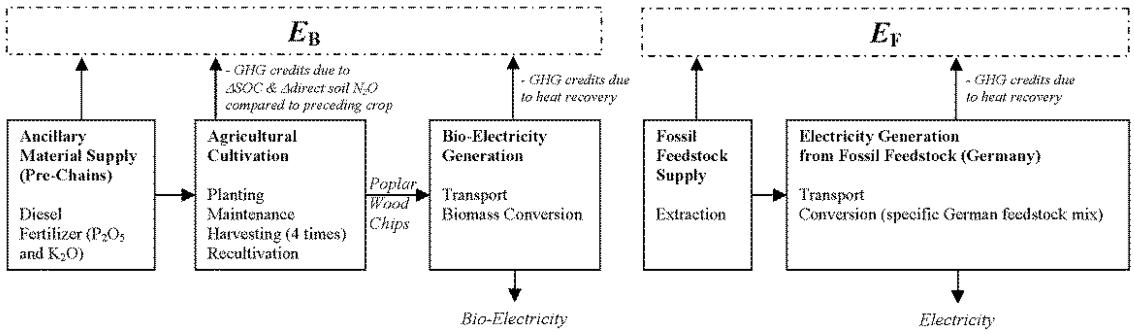

Figure 6.1: Schematic of processes under consideration for the calculation of the cumulative greenhouse gas emissions (comprising CO_2, CH_4 and N_2O) from a second generation bio-electricity production system from poplar wood chips gasification (E_B) as well as from a fossil reference system (E_F)

We used the LCA software umberto® [33] to define the overall electricity generation system from short rotation poplar cultivation under East German conditions to calculate the accompanied GHG emissions E_B and to carry out the MC calculations for the parameters under study.

The relevance for the variance of the GHG mitigation of bioelectricity of the different parameters was obtained with a Spearman rank correlation analysis using SAS 9.2 [34].

GHG emissions due to SRC cultivation and biomass conversion (E_B)

- *SRC cultivation*

The emissions from pre-chains, i.e. the production of diesel fuel for farming operations and the transport of wood chips as well as the production of combined phosphorous and potassium fertiliser, are considered for E_B . The direct emissions from the tractor used for the different farming operations were included, according to diesel consumption values stated in Ref. [35], which sum up to 548 L ha^{-1} diesel for the complete plantation standing time (details in Table S1[7]). A 4-year harvest cycle of the wood chips and a total plantation standing time of 16 years was assumed as economically reasonable under German cultivation conditions [36]. Emissions from nitrogen fertilisers were not considered since they are commonly not applied on commercial poplar plantations in Germany [37]. On the contrary, N fertilisation was found to be responsible for increased weed growth as well as for nitrate leaching and, on the other hand, did not affect wood yield [38].

Additional effects from land use change, such as changes in carbon stocks in vegetation as well as soils, are taken into account. The resulting GHG emissions are regarded as important drivers if bioenergy can be considered as potentially GHG-mitigating or not (e.g. Refs. [39,40]). This direct LUC is considered in the European and national frameworks for GHG calculations [7,10,11], whereas indirect land use change (iLUC) effects are being considered only quite recently [41]. In the specific context of this study, iLUC was not taken into account, supposing a negligible effect of substituting rye production with SRC:

In Brandenburg, there are several principal options where one can establish SRC: degraded land, set-aside land, abandoned grassland or low-yielding arable land. Degraded lands, often recommended for bioenergy production [42-44], are rare in Germany. Attempts for planting black locust (*Robinia pseudoacacia*) SRC in the large former open-cast coal mining areas are still in the experimental stage [45]. Plantations on set-aside land, as assumed by, e.g. Ref. [46], are not feasible in Germany any more because of the decreasing availability of set-aside land (28 % less acreage in 2011 compared to 2000 [47]). Permanent grasslands are preserved by legal regulations in several German federal states where grassland losses of already >5% compared to the 2005 baseline were observed [48-50]. For that reason, grassland, which is not suitable for animal feed, is hardly used for the establishment of plantations, but is rather used directly as biomass supply for anaerobic digestion [51] or combustion [52]. Nevertheless, a comprehensive study indicated that from an economic point of view as well as with regard to GHG mitigation, the SRC establishment on grassland might be favourable to direct grassland biomass use [53]. The present situation in the federal state of Brandenburg seems to be that farmers establish new plantations on low-yielding agricultural land that is distant from the main farm areas [37,54]. Hence, we assumed rye (*Secale cereale*) as the preceding crop, which is the predominant crop on less fertile soils in Brandenburg.

Direct N_2O emissions from soils and soil carbon stock changes are taken into account in this study as the resulting effects of direct LUC from rye to poplar. The difference between measured N_2O emissions under altered land use is not yet considered in the common frameworks [7]. N_2O emissions are usually calculated depending on anthropogenic N inputs (e.g. fertilisation levels). N_2O savings due to LUC are not considered albeit N_2O has a great influence on the GHG balance due to its high global warming potential and albeit savings are considered for changed soil C stocks. For this case study, direct N_2O emission values from the plantation site are available from measurements [55]. From the unfertilised poplar experimental stand in Brandenburg, N_2O emissions were found to be

[7] This online supplementary data Table S1 is also available as Table 12.1 in 12.2.

(0.79 ± 0.18) kg N_2O $ha^{-1}yr^{-1}$ (mean ± SD), which is in the same order of magnitude as in several reference studies [55]. An unfertilised rye plot, comparable in soil type and climatic conditions, was found to emit (1.57 ± 0.32) kg N_2O ha^{-1} yr^{-1}. We suppose that the resulting credits are only applicable during the first 16 years of bio-electricity generation with poplar wood chips. If afterwards a plantation is re-grown on the same plot, no N_2O credits can be given to the bio-electricity. The carbon stock under poplar is supposed to increase compared to former cropland with 27,500 kg CO_{2e} ha^{-1} for a time frame of 20 years [56]. This value was annualised for the plantation standing time of 16 years and assumed for this study (1719 kg CO_{2e} ha^{-1} yr^{-1}).

- *Biomass conversion*

The specific values for the second generation biomass conversion process were derived from a facility in southern Germany, where fluidised bed gasifiers with a downstream organic Rankine cycle are currently being installed [57,58] (cf.Table 6.1).

No direct CO_2 emissions from the conversion site itself were considered under the assumption that biomass is converted to energy without further input of fossil based energy and that the released biogenic carbon will be reabsorbed again from vegetation.

Evans et al. [44] reviewed several studies which reported a wide range of conversion efficiencies of biomass gasification processes (mean ± SD;(30 ± 2)%).The expected efficiency of 33 % for the process investigated here lies in the upper range [57].

The facility will provide electric as well as thermal energy und thus gain a high degree of an expected overall efficiency of approximately 80 % [58]. The GHG emissions (respective credits) were therefore allocated to the electricity according to the output ratio (c.f.Table 6.1) of electric and thermal energy with approx. 44 vs. 56 %. This provides an appropriate comparator value of E_B vs. E_F , the latter being derived also according to an allocation approach.

Table 6.1: Assumed characteristics of the gasification process [57]

	Unit	Value
Feedstock (wood chips $_{wet}$)	t yr^{-1}	45,000
Moisture content of wood chips	%	50
Lower heating value of absolutely dry wood	MJ kg_{DM}^{-1}	18.3
Lower heating value of wood chips at 50 % moisture	MJ kg_{FM}^{-1}	7.93
Mean transport distance of wood chips	km	30
Operating hours	h yr^{-1}	7,000
Installed electric capacity of combined heat and power units	MW	4.5
Installed electric capacity of organic Rankine cycle	MW	0.5
Installed thermal capacity of combined heat and power units	MW	6.4
Conversion efficiency	%	33
Electricity generation	MWh yr^{-1}	35,000

GHG emissions of the conventional electricity generation in Germany (E_F)

The reference E_F for the displaced fossil electricity generation system in Germany in this study is based on the work of Ref. [59] who modelled the entire German power generation mix, taking into account each generation facility in Germany for the years 2006 and 2007. For each of the renewable energy sources, the substituted fossil mix was identified according to their specific feed-in characteristics, the merit-order effects at the European Energy Exchange and also shut-down times of nuclear power plants. Their analyses for 2006 and 2007 included only the direct but not yet pre-chain CO_{2e} emissions. The German Federal Environmental Agency (UBA) is updating yearly the emission balance of renewable energies in Germany and bases its calculations on the analyses of Ref. [59]. Since 2009, pre-chains are included in their emission balances and were considered here (c.f. Table 6.2).

Differences in the emission factors over the years are due to higher conversion efficiencies of the power plants as well as to the replacement of old brown coal plants and reduced hard coal use. The differences in the feedstock fractions in 2006 compared to 2007 stem from nuclear power plant shut-downs in 2007, low CO_2 certificate prices and the use of brown coal plants as base load instead of middle load plants. Thus, the brown coal plants were not substituted by solid biomass in the usual amount.

Even though the share of renewable energies within the total energy generation mix has increased during recent years and thus, the emission factor of electric energy should have decreased remarkably, the overall increase in electricity demand counteracts this decrease.

In this study, we chose the latest available emission factor for 2009 as E_F (0.236 kg CO_{2e} MJ^{-1}) that already includes prechain emissions from the respective fossil feedstock extraction and processing. It represents the share of emissions produced by the German power generation facilities already allocated to the electric energy.

Table 6.2: Fractions of substituted fossil feedstock for electricity generation in Germany through solid biomass (%) and the feedstock specific and aggregated emission factors in 2006, 2007 (excluding pre-chains) and 2007, 2009 (including pre-chains) (kg CO_2e MJ^{-1}) [59-61]

Substituted feedstock	in %		Emission Factor (feedstock specific) kg MJ^{-1}	
	2006	2007	2007[a]	2009[a, b]
Brown coal	16	2	0.304[a]	0.306[a]
Hard coal	59	73	0.278[a]	0.266[a]
Natural Gas	25	25	0.148[a]	0.122[a]
Emission Factor E_F (aggregated)			0.246[a]	0.236[a]

[a] includes pre-chain emissions, [b] substituted feedstock fractions as in 2006

Uncertainty analyses

- *Monte Carlo simulations*

To calculate the complete CO_{2e} mitigation potential of the specific second generation conversion pathway, many parameters have to be defined. Some of them are uncertain, e.g. the conversion efficiency or transport distances. Some are variable since we study a natural system with naturally varying flows, e.g. N_2O emissions from soil or wood yields. Other parameters address system assumptions, for example allocation rules or LUC matters. To deal with these uncertainties and the variability of the system parameters, Monte Carlo simulations with 5000 samples were carried out. For each of the system parameters as given in Table 6.3, probability density functions were assigned and the effect on the overall mitigation potential of second generation bioelectricity (kg CO_{2e} MJ^{-1}) was observed. If detailed information on the variable was known, the appropriate probability density function was assigned. This is the case for the soil N_2O emissions from the poplar plantation as well as for the soil N_2O emissions from the unfertilised rye, where normal distributions with N(mean, standard deviation) were assigned. If no specific information about the distribution was available, variables were drawn from rectangular distributions with R(lower boundary, upper boundary).

Due to the fact that the global warming potentials are repeatedly under investigation [62] and modification [32], they were also varied within the Monte Carlo analysis. If direct and indirect radiative effects of aerosol responses in the atmosphere are included, a maximum GWP = 40 for methane was reported [62]. The different estimates for the GWPs of methane and nitrous oxide documented in the 2nd and 4th IPCC assessment reports were also accounted for in the MC simulations.

The assumptions regarding the variance of the German fossil reference emission values E_F between (0.236 and 0.246) kg CO_{2e} MJ^{-1} were already discussed in Section 6.1.2 (*GHG emissions of the conventional electricity…*).

The allocation approach was varied in the MC calculations between no allocations of the heat output at all and the 56 % derived from Table 6.1. Instead of these energy yield based shares of electricity and heat, an economic allocation according to the revenues would result approximately in a splitting of 66 vs. -34 %. This is already integrated in the MC range for the allocation parameter.

Besides the full MC parameter set, we considered three additional scenarios. They indicate the importance of the two LUC system assumptions, i.e. that SOC content is increased and a N_2O emission reduction is possible. One minimum scenario without LUC (neither credits for SOC increase nor for avoided N_2O emissions from rye cultivation are given) as well as two mixed scenarios which assume either SOC increase or N_2O reductions are calculated. The remaining parameters were varied according to Table 6.3.

Table 6.3: Parameters for the case study site and the parameter-set varied within Monte Carlo (MC) simulation (assigned distributions, literature references)

Parameters under study within MC simulation	Site-specific values	Unit	Distribution specifications	Remarks	Reference
Soil N_2O emissions from unfertilized poplar plantations on sandy soils	0.79	kg N_2O $ha^{-1}yr^{-1}$	N(0.79, 0.18)	Specification taken from long-term field studies	[55]
Soil N_2O emissions from unfertilized rye plots on sandy soils	1.57	kg N_2O $ha^{-1}yr^{-1}$	N(1.57, 0.32)	Specification taken from long-term field studies	[55]
Allocation of heat extraction	0.56	%	R(0, 0.56)	Range between no and partial allocation of the heat extraction	Own assumptions, [58]
Transport distance – plantation to gasification site	30	km	R(10, 100)	Representation of regional supply	Own assumptions
Electric conversion efficiency	33	%	N(30, 2)	Actual range of conversion efficiencies in wood gasification plants	[44,58]
Woodchips yield $_{wet}$	14.5	10^3 kg ha^{-1}	R(8.6, 20)	Range between low and high yielding areas	[35]
Soil organic carbon change (sink)	-1 719	kg CO_{2e} $ha^{-1}yr^{-1}$	R(-1 719, 0)	Range between no and complete consideration of carbon credits for soil carbon changes on cropland (rye) changed to SRC	[56]
GHG emissions of reference conventional electricity generation in Germany	0.235	kg CO_{2e} MJ^{-1}	R(0.236, 0.246)	Emission Factor for grid electricity (excluding biomass share)	[59,60,63]
GWP (100) N_2O	298	kg CO_{2e}	R(298, 310)	Range of GWP for 100-yr time horizon between IPCC 2nd and 4th Assessment Report	[32]
GWP (100) CH_4	25	kg CO_{2e}	R(21, 40)	Min. GWP for 100-yr time horizon from IPCC 2nd Assessment Report, Max. from including direct and indirect aerosol effects from [51]	[32,62]

GHG - Greenhouse Gas; GWP - Global Warming Potential; SRC - Short Rotation Coppice; Rectangular distribution R(lower boundary, upper boundary); Normal distribution N(mean, standard deviation)

- *Minimum/maximum analyses*

A usual way of performing an uncertainty analysis is a Minimum/Maximum analysis. All parameters are set to their assumed minimum and maximum values, respectively, and the resulting emission range indicates the possible outcome of the study. In the complex system here, this approach was modified to account for that some of the parameters are indicating credits, for example the allocation assumption for heat recovery or reduced N_2O emissions from land use change (poplar vs. rye). Otherwise the minimum parameter set would not yield the minimum emissions because the credits were also minimised. Therefore, our parameter sets consisted of those parameter combinations that would generate a maximum as well as a minimum emission value E_B and MF_B value, respectively.

6.1.3 Results

Specific emission factor E_B for bio-electricity from SRC

We obtained gross GHG emissions E_B = 0.012 kg CO_{2e} MJ^{-1} for the case study bio-electricity generation system (parameter set is provided in Table 6.3). The main contributor to these GHG emissions was N_2O, followed by CO_2 whereas CH_4 was of minor importance (Table 6.4). The N_2O emissions from the soil dominated the balance (58 %). Of all the agricultural processes, harvesting contributed most to the gross emissions (20 %), planting and recultivation of the site were less important (<2 %). The pre-chains and transport emitted similar amounts. After the credits for SOC accumulation, for N_2O emission reduction and for heat recovery were given, the case study had negative net GHG emissions E_B = -0.059 kg CO_{2e} MJ^{-1}.

Table 6.4: Relative contribution [%] of greenhouse gas emissions from biomass production to the total CO_{2e} emissions for the case study [kg CO_{2e} MJ^{-1}] before credits are given (i) for soil organic carbon accumulation, (ii) for N_2O emissions savings vs. reference crop rye and (iii) for heat recovery

	Relative contribution [%]			Total emissions	
Greenhouse Gas Process Level	CH_4	CO_2	N_2O	[%]	[kg CO_{2e} MJ^{-1}]
Pre-Chains	1.2	10.8	0.1	12.1	0.0015
Plantation Planting	<0.01	1.4	<0.01	1.4	0.0002
Harvesting	<0.01	19.6	0.3	19.9	0.0024
Recultivation	<0.01	1.1	<0.01	1.1	0.0001
Soil	*n/a*	*n/a*	57.5	57.5	0.0069
Transport	<0.01	7.7	0.2	7.9	0.0010
Total emissions [%] [a]	1.2	40.7	58.2	100.0	
[kg CO_{2e} MJ^{-1}]	0.0001	0.0049	0.0070		0.0120
				Credits	
Soil carbon accumulation					- 0.0503
N_2O from rye					- 0.0137
Heat recovery					- 0.0067

[a] Differences due to rounding possible; *n/a* not available

The credits which were given for avoided N_2O emissions from rye cultivation overcompensated the N_2O emissions from the poplar site soil by nearly 100 % and contributed to the climate change mitigating status of the bio-electricity. Even if no N_2O benefits were assumed, the bio-electricity still mitigates CO_{2e} as long as it is assumed that the SOC content increases (Table 6.5). From the complete MC parameter set (cf. Table 6.3), we got net GHG reductions E_B = (-0.034 ± 0.021)kg CO_{2e} MJ^{-1} (median ± SD), i.e. less greenhouse gases are emitted than produced (Table 6.5).

Table 6.5: Scenario results for net E_B (top value in each cell) and MF_B (bottom value in each cell) from MC analyses [median ± SD; kg CO_{2e} MJ^{-1}]

Credits for LUC aspects		N_2O emission reduction [kg CO_{2e} MJ^{-1}]	
		0	- 0.0137
SOC increase [kg CO_{2e} MJ^{-1}]	0	0.010 ± 0.004 0.230 ± 0.005	- 0.005 ± 0.005 0.246 ± 0.006
	- 0.0503	- 0.018 ± 0.019 0.259 ± 0.019	- 0.034 ± 0.021 0.274 ± 0.021

The uncertainty regarding increasing SOC contributes more to the overall variability of E_B than that of the N_2O emissions from the reference crop rye, similar to the overall variability of MF_B (Table 6.6; see Section 3.3[8]).

The Min/Max analysis indicated more conservative emission assumptions, independent of LUC aspects (mean values for no credits/credits given E_B 0.016/-0.019 kg CO_{2e} MJ^{-1}).

Greenhouse gas mitigation

The overall mitigation E_F -E_B (or MF_B) of the second generation pathway case study resulted in 0.294 kg CO_{2e} MJ^{-1}, i.e. electricity could be generated with fewer GHG emissions than if it had been generated from the fossil reference system.

From the MC calculation MF B was identified as (0.274 ± 0.021) kg CO_{2e} MJ^{-1} for the full parameter set. The case-study's mitigation factor is near the MC simulations upper limit, indicating that its SOC parameter represents the upper range of the assumed parameter in the MC parameter set (Figure 6.2).

In line with the GHG emissions E_B, the mitigation factors by the Min/Max analyses were also indicated with lower mean values (MF_B = 0.224 and 0.261 kg CO_{2e} MJ^{-1}) than from the MC simulations.

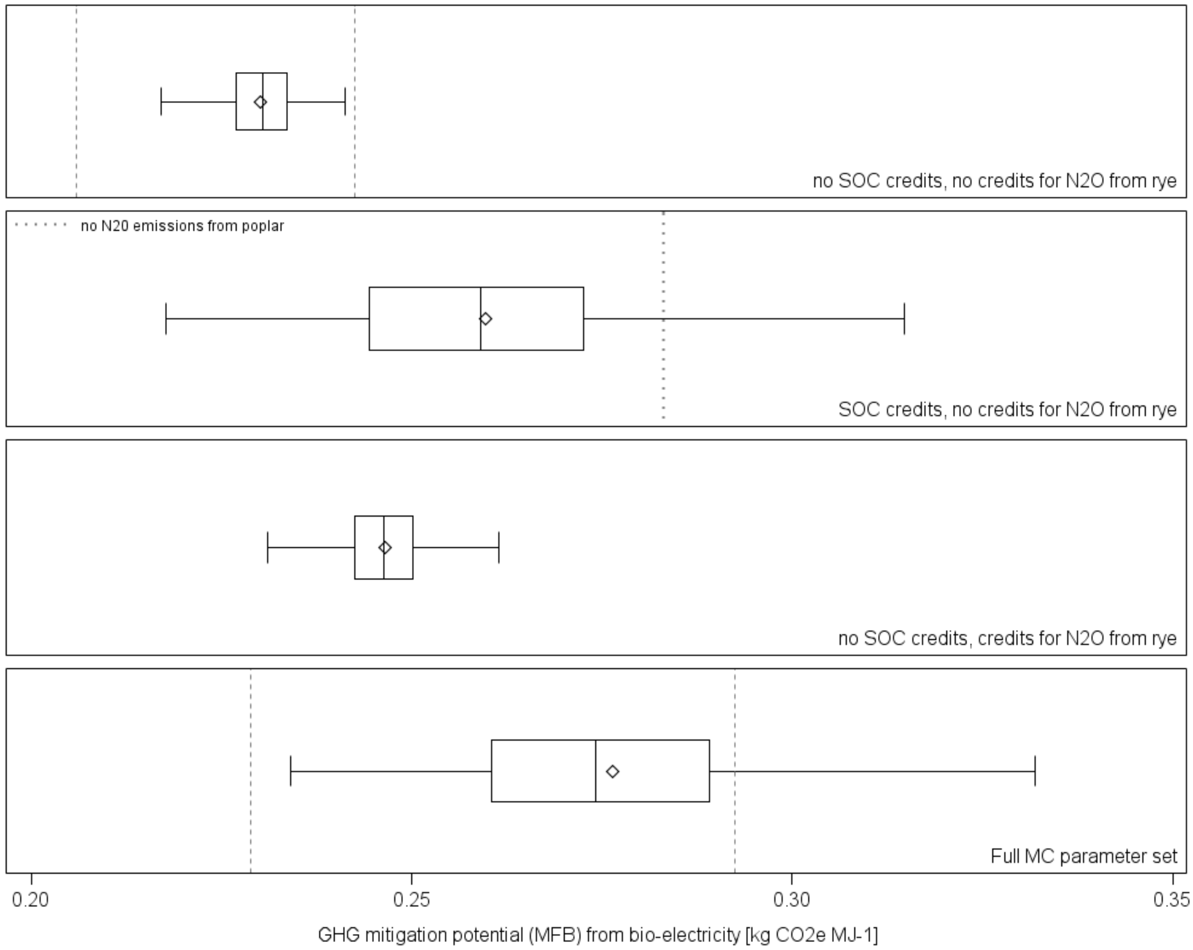

Figure 6.2: GHG mitigation factors *MF_B* from second generation bio-electricity production from poplar SRC for the complete MC parameter set and for the scenario-based analyses ([kg CO_{2e} MJ^{-1}]; boxes represent interquartile ranges between 25th and 75th Quartile, whiskers indicate minimum and maximum values respectively within 1.5 of IQR; '◊' marks the mean value and the solid line the median value; dashed lines mark the maximum and minimum GHG emissions in the Min/Max-based uncertainty analysis)

[8] Refers to sub-section "*Parameter importance for the variance of the mitigation potential*" below

Parameter importance for the variance of the mitigation potential

As already indicated from the aggregated MC results in the scenario analyses (see sub-sections in 6.1.3: *Specific emission factor E_B and Greenhouse gas mitigation*), some parameters contribute more than others to the uncertainty of the estimated GHG emissions and the mitigation effect, respectively.

From a Spearman rank correlation analysis, the relative relevance of the parameters that address uncertainty was obtained in the following order (Table 6.6): for the common system approach which assumes SOC increases under SRC plantations (scenario (1;0)), the variability of this parameter was the most important and a further small contribution was made by the variance of the yields. The heat allocation, the fossil reference system and the direct N_2O emissions from the poplar site were of minor importance for the overall variability of the mitigation factor. The uncertainty within those parameters that address uncertainties in the global warming potentials of nitrous oxide and methane were not relevant (all $p > 0.0001$; thus not included in Table 6.6).

If the land use change aspects (credits for SOC increase and N_2O avoidance) were assessed within the MC calculations (scenario: full MC Set (1;1)), the uncertainty of the SOC accumulation had the most prominent effect on the variance of the overall mitigation potential. The next important parameter variability was the variance of yields.

If LUC effects were ignored (scenario: 0;0), the uncertainty of the fossil reference system emissions was the most contributing parameter, followed by the variability of heat recovery and yields.

Table 6.6: Relative contribution [%] of the different MC parameters (given in Table 6.3) to the overall uncertainty of the GHG mitigation of bio-electricity from gasified SRC wood chips (for the complete MC parameter set and three scenarios: credits for soil organic carbon sink and N_2O reference emissions ignored as well as partly considered)

Parameters considered in Monte Carlo analysis	Relative Contribution [%]			
	Full MC Set (1;1)	SOC/N_2O (0;0)	SOC/N_2O (1;0)	SOC/N_2O (0;1)
N_2O ref. emissions from unfertilized rye on sandy soils	2.2	-	-	**34.4**
N_2O emissions from unfertilized poplar on sandy soils	0.4	7.7	0.6	5.3
Woodchips yield$_{wet}$	15.5	19.4	5.2	9
Transport distance – plantation to gasification site	*n/a*	3.3	*n/a*	1.9
Soil organic carbon change (sink)	**78.6**	-	**90**	-
Allocation of heat extraction	1.6	27.1	1.6	20.8
Electric conversion efficiency	0.6	1.3	*n/a*	0.5
GHG emissions of German reference electricity generation	1.2	**41.2**	2.1	28.1

n/a indicates that the parameter uncertainty was not contributing to the mitigation uncertainty, $p > 0.0001$

6.1.4 Discussion

The net emissions E_B from bio-electricity generated via the second generation pathway were lower than those found in previous gasification studies (Table 6.7). They were even lower compared to other renewable non-biomass energy sources, whereas the mitigation factor MF_B was found to be in the same range.

Wood chip conversion in combined heat and power plants as a different conversion pathway was found to be associated with higher emissions. However, these studies did not consider credits for soil organic carbon change [46,60]. Only one study also calculated credits (E_B = -0.342 kg CO_{2e} MJ^{-1}), resulting in a comparably high mitigation for the bioelectricity (MF_B = 0.516 kg CO_{2e} MJ^{-1}), whereas Refs. [64,65] identified noticeably lower mitigation effects for the gasification pathway. This could be assigned to the fact that willows which are cultivated as SRC are often fertilised as in Ref. [64].

The relative variability (coefficient of variance) of the net GHG emissions E_B was high (62 %) compared to the relative overall variability of the mitigation factor MF_B (7 %). This was due to the low variability of the reference emission factor E_F for which detailed information was available. In absolute terms, however, the mitigation effect and its variance are determined by the variance of the reference emission factor E_F.

Table 6.7: Published net GHG emissions (E_B) and mitigation factors (MF_B) for different renewable conversion pathways compared to the results from this study [kg CO_2e MJ^{-1}]

Biomass conversion pathway	Feedstock	E_B incl. pre-chains [kg CO_2e MJ^{-1}]	MF_B [kg CO_2e MJ^{-1}]	Reference
Average (literature review)		0.017	*n/a*	[44]
CHP	poplar	0.006 [a]	*n/a*	[46]
CHP	solid biomass	0.005 [a]	*n/a*	[60]
CHP	wood chips	-0.342	0.516	[66]
	wood chips	0.003 [a]		[13]
		0.004 [a]		[13]
Gasification (small-scale, with ORC)	willow	0.018 [a, b]	*n/a*	[67]
Gasification	willow chips	*n/a*	0.062-0.089	[64]
Gasification	woody biomass	0.009	*n/a*	[68]
Gasification	poplar chips	-0.034±0.021	0.274±0.021	this study
Gasification [a]	poplar chips	-0.005±0.005	0.246±0.006	this study
Biomethane-gas-steam power plant	wood chips	*n/a*	-0.030-+0.065	[65]
Renewable energy sources (non-biomass) (tidal, wind geothermal)		0.001 – 0.002	*n/a*	[69]
		n/a	0.217-0.246	[59]

[a] no soil organic carbon changes considered; [b] only CO_2 emissions considered; CHP combined heat and power; ORC organic Rankine cycle; *n/a* not available

The range of the results for E_B and MF_B varies according to the uncertainty associated with the variables. In this case study, some parameters (like baseline N_2O emissions for rye and poplar) were relatively certain due to intensive measurements, while to others (carbon sequestration due to land use change) a high variability due to lack of knowledge was assigned. In different environments with different availability of knowledge other processes may dominate the variability of the results. The variability of the emissions due to the electricity generation from biomass is very low compared to other studies, which were based on common default ranges of emissions. Meyer-Aurich et al. [70] found for example a range from 0.1 to 0.4 kg CO_{2e} kWh^{-1} (0.36-1.44 kg CO_{2e} MJ^{-1}) emissions associated with the production of bioenergy from agricultural feedstocks via biogas. This is more than 10 times the range that we found for this case study. Reasons are, for example, a much higher variability of emissions which are associated with the usable feedstock for biogas production (silage corn, cattle slurry) or with the management variants (e.g. digestate handling). This indicates that for specific bioenergy production systems the uncertainty regarding the GHG emissions may be substantially lower than that calculated for a greater range of settings.

Both emission factors, from poplar as well from rye, had a small variability on the case study site (±0.18 kg N_2O $ha^{-1}yr^{-1}$ and ±0.32 kg N_2O $ha^{-1}yr^{-1}$), whereas recent measurements, for example in an unmanaged beech forest in Germany, indicated a higher background variability of (0.77 ± 0.69) kg CO_{2e} ha^{-1} yr^{-1} [71]. Hence, the accounting for the uncertainty in soil N_2O emissions from the reference agricultural system did not contribute much to the overall variability of E_B and MF_B , as the Spearman rank correlation indicated. Even though the small variance of the N_2O emission factors is based on measurements on a few plots only, also other authors stated such differences between SRC poplar and annual crops (significant results) [72]. New data from an adjacent experimental site with a randomised design indicated that the absolute N_2O emissions from poplar assumed for this study are appropriate [38].

However, baseline emissions are available only in exceptional cases for different agricultural crops compared to SRC with the associated probability distribution functions. Instead modellers and policy makers are forced to use the parameters and their ranges provided for example by IPCC. This results in a higher contribution of the variability of the process to the overall variability of the outcome. Another reason for the little contribution of the variability of N_2O to the variability of the results is that no nitrogen fertiliser has been used. In bioenergy systems, which rely on nitrogen fertiliser application to energy crops, both the total GHG emissions due to N_2O emissions and the relative contribution of the uncertainty range to the overall variability of the results is much higher, as for example in the biogas systems mentioned above or Refs. [25,70].

The integration of N_2O credits due to changed land management was a newly applied system assumption. The N_2O emissions from the poplar site were responsible for more than half of the GHG emissions due to the bio-electricity generation (Table 6.4). If these GHG emissions from poplar SRC were taken out of the balance as well as the credits for the avoided N_2O emissions from the reference crop rye, the resulting mitigation factor (MF_B = 0.283 kg CO_{2e} MJ^{-1}) would be still in the MC range (Figure 6.2). SRC becomes the new baseline if poplar is regrown after 16 years on the same plot. Which land use option the farmers actually choose after the poplars are grubbed, still needs further investigation. Today, hardly any plantations in Germany have reached this age.

As in numerous previous studies [73], the integration of soil organic carbon changes had a substantial influence on the outcome of the net emission calculations. This case study indicates once more that reliable information about carbon sequestration under SRC can contribute to improved mitigation assessments of bioenergy systems [74-76].

SOC changes are also the main aspect within the discussion of indirect land use change effects which are already reviewed in detail by other authors, e.g. Ref. [77]. Just recently, a proposal was presented to the public which amended the GHG accountings calculated after RED with iLUC factors for different biofuel feedstocks [41]. It can be assumed that such iLUC factors will also be introduced for other biomass-for-energy-uses than biofuels/bioliquids with special consideration of forestry aspects, for example regarding allocation time frames. The uncertainty within these iLUC factors could also easily be represented in the mitigation calculations via MC simulations. In the future, indirect LUC could become more important in the system assessed here: if in the future more farmers in Germany decide to establish SRC and replace their food/feed crops, relevant amounts of food/feed would be missing on the market. This, in turn, may induce intensification, or for example recultivation of abandoned land in Eastern Europe or Ukraine [78].

6.1.5 Conclusions

The second generation bio-electricity generated from poplar wood chips under German conditions was found to enable climate change mitigation MF_B = 0.294 kg CO_{2e} MJ^{-1} in this specific case study. If uncertainties from parameters, for example from yield, from N_2O emissions or from soil organic carbon changes, were included using Monte Carlo simulations, the simulated mitigation was (0.274 ± 0.021) kg CO_{2e} MJ^{-1} indicating a comparatively low relative variability (7 %). Mitigating greenhouse gas emissions via bioelectricity generation from wood chip gasification is a certain option, if happening under this specific site (low and certain background N_2O emissions) and management conditions (no nitrogen fertiliser). However, results are not to be transferred to other bioenergy systems but have to be assessed instead system by system, because contributions to uncertainty from other system processes may be completely different.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.biombioe.2013.05.004[9]

[9] Table S1 also available as Table 12.1 in the Supplement (12.2)

6.1.6 References

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6.2 Resource Use in the Context of Climate Change Mitigation - Effects of Complexity and Uncertainty of Agricultural Products for Multi-criteria Assessment of Systems

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6.2.1 Introduction

Need for Resource Usage Strategies

Bioresources are increasingly substituted for limited fossil resources, such as crude oil or natural gas, for energy generation as well as for the provision of industrial raw materials such as plastics [1] or industrial fiber [2]. One of the main drivers for this substitution—besides limited availability and the resulting need for sustainable depletion—is the climate impact of their use. However, fossil resources are still essential as raw materials in the chemical industry, even though globally only 10 % of them are used as such [3]. To meet growing bioresource demand, cropland is required which is also a limited resource [4]. Land under crop cultivation was explicitly termed a planetary boundary that needs careful management [5]. Therefore, the trade-offs of different usage strategies for these limited resources should be carefully assessed [6].

Material Alternatives for Building Insulations

In general, insulation materials can be derived from several sources: from renewable bioresources (such as plant fibers grown on cropland), from inorganic resources (like mineral fibers produced from rocks), as well as from fossil resources (like synthetic materials, for example polystyrene). Mineral and synthetic insulation products dominate the insulation market [7] (e.g., 96 % in Germany), with expanded polystyrene (EPS) being the second-most common (32 %) after mineral rock wool (51 %). Materials from bioresources represent 4 %, of which approx. 10 % are from hemp (*Cannabis sativa* L.) and flax (*Linum usitatissimum* L.) [8].

Assessments of the Environmental Impacts of Building Insulations

System analysis often uses Life Cycle Assessment (LCA) based approaches to analyze different environmental impacts and to address environmental sustainability issues. For adding an insulation to a building, Erlandsson et al. [9] used such an approach: They found that the additional environmental effects from insulation production and installation, for example GHG emissions, were more than balanced by the emission savings achieved by the insulation throughout its entire service life. Subsequent comparative studies between insulation materials confirmed that the differences between them, stemming from their production processes, were of minor significance due to the overwhelming impact of the life cycle stage “use in building” [10]. The latter created energy savings of about 100 times the energy used for production and disposal [10]. Accordingly, insulating buildings is an appropriate way to mitigate climate change. However, we believe that it should be assessed which insulation material generates as low environmental impacts as possible if limitations regarding available land and fossil resources are considered.

Since 2008, environmental product declarations, for instance from BBSR [11], provide values for environmental impact categories that are common in LCA: for example climate change, acidification, or ozone depletion. They also include the total energy demand (fossil/renewable); end-of-life credits are sometimes accounted for as well. Usually, the resource demand for land is not included as a separate impact category. Impacts are typically expressed “per mass” or “per volume” of the respective insulation material. However, several authors have stressed that comparisons between different bioresource products should use the necessary cultivation area as a comparison basis instead of a mass-based one (for example for bio-based polymers [12], or biofuels [13,14]). If fossil-based products are to be compared with bio-based ones, or in complex product/service systems, other functional units can be more appropriate [15]. Nevertheless, land use needs to be considered in the evaluation.

Objectives

Our main objective is to analyze usage strategies for the limited bioresource cropland and its derived products and the limited resource fossil fuels, as well as the trade-offs. As a result, we will attempt to identify the most beneficial combination of the resource use with regard to GHG mitigation. To achieve this, we assessed two alternative strategies with an LCA-based approach:

- *Biomateria*l strategy: Cropland is used to grow fiber plants as bioresources, which are further processed to an insulation material with fossil energy derived from crude oil and natural gas; or
- *Bioenergy* strategy: Cropland is used to grow energy crops (short rotation coppice or maize (*Zea mays* L.) as bioresources and the bioenergy is then used for the production of fossil fuel-based synthetic insulation materials.

6.2.2 Materials and Methods

Methodological Approach: Life Cycle Assessment

- *General Method*

LCA is a method to describe the environmental impacts of products or services, taking their whole life cycle into account. The impacts are expressed for an equivalent functional unit (FU) so that systems perform equally. Methodological LCA standards exist [16,17]. Social and economic sustainability are not yet fully implemented in the methodology (social Life Cycle Assessment, SLCA; Life Cycle Costing, LCC) [18,19]. One important methodological choice is how to deal with the multi-functionality of systems. This analysis followed a substitution approach to account for co-product generation of the two resource usage strategies (Figure 6.3).

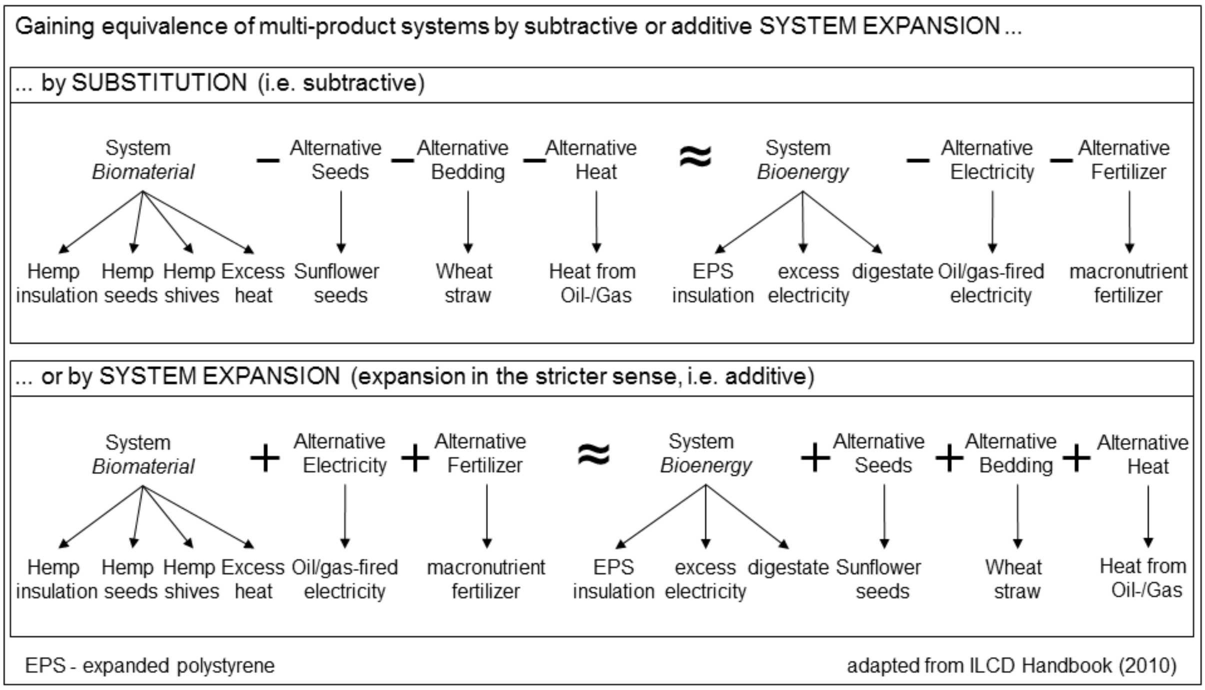

Figure 6.3: Two options to make the multi-product systems comparable by system expansion. This study followed the upper approach, i.e., subtracting of alternative co-products (graph adapted from ILCD Handbook [20])

This can be regarded as a variant of system expansion [20] (pp. 77–79). No explicit economic models were used to investigate market effects in the two bioresource strategies, but we contacted experts for their opinion on the substitution options. The limitations of this approach, such as an inherent uncertainty of effects of changes and a risk of unfair results, were discussed in Ekvall et al. [21], and with a focus on bio-based materials in Pawelzik et al. [22].

- *System Boundaries*

Two complex systems were modeled to represent the two resource usage and mitigation strategies using building insulation (Figure 6.4). They included the complete life cycle of building insulation from raw material extraction through insulation production to their end of life. The time at the building ("usage phase") was excluded from this analysis due to the following reasons: (i) The construction characteristics of the insulation materials are the same; (ii) the period during which they are attached to a building can be assumed to be 40 to 50 years for both [23]. During this period, they provided the same insulation effect, which can be represented by specific heat transfer coefficients (see also definition of functional unit in Section 2.1.3). We assumed that detached insulation materials at their end of life would be co-incinerated instead of re-used, as re-use has not yet been widely adopted due to time-consuming, costly processes.

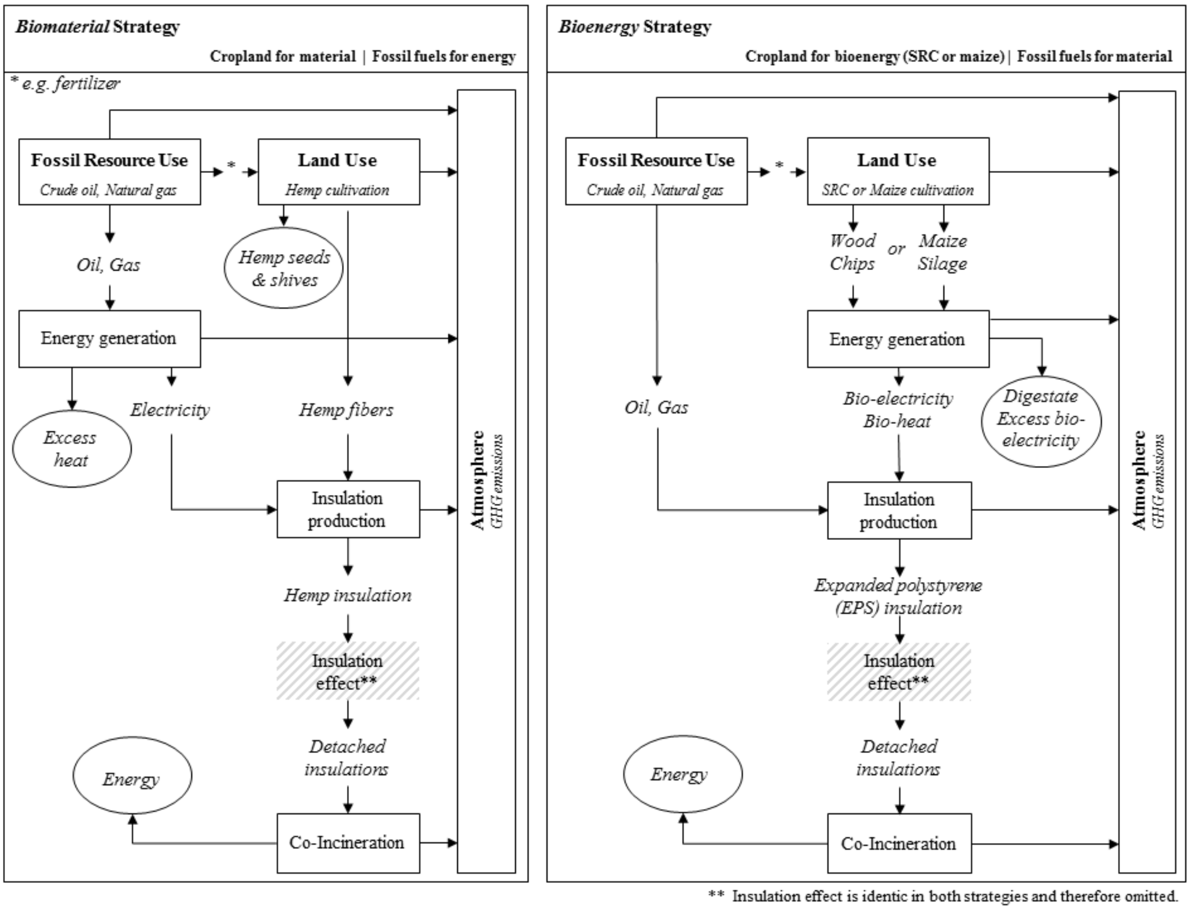

Figure 6.4: Two strategies to achieve lower GHG emissions by combined use of cropland and fossil resources: cropland for material production (fibers; *biomaterial* strategy; left) or for bioenergy (short rotation coppice, maize; *bioenergy* strategy; right), and fossil fuels for energy generation or for material production (synthetic foam) using the example of two building insulations either made of hemp fiber (*Cannabis sativa* L.) or of expanded polystyrene. Co-products are indicated by oval frames

For our *biomaterial* strategy, we chose to represent it with a system that used the bioresource fiber hemp (*Cannabis sativa* L.) as a material for the insulation. Hemp-based materials for building applications have increasingly raised interest, as suggested by a growing number of publications [24]. Oil- and gas-fired co-generation plants provided the fossil energy. This choice of energy technology made it comparable: It arose from the type of fossil fuel resources, which were processed for the synthetic insulation in the alternative system. Hemp seeds, shives, and excess heat arose as co-products.

In our *bioenergy* strategy, the synthetic insulation material EPS was produced from the fossil resources crude oil and natural gas. The cropland provided bioenergy feedstock as bioresources: either wood chips from poplar (*Populus* spp.) in short-rotation coppice (SRC), which could be gasified (*bioenergy* option SRC), or silage from maize (*Zea mays* L.), which could be digested to biogas (*bioenergy* option maize). These bioenergy technologies represent two common pathways [25] (p. 5). Excess electricity and—if maize silage is used as feedstock—biogas digestate arose as co-products in the *bioenergy* strategy.

- *Functional Unit*

We defined the FU as one square meter of insulation material ($A = 1\ m^2$) with the *specific heat transfer coefficient* $U = 0.2\ W \cdot m^{-2} \cdot K^{-1}$ [26]. We assumed that the insulation materials are installed on an existing building wall and covered by plaster. Accordingly, inner and outer thermal resistances are equal for both materials. As a consequence, using the Equation (2.1):

$$L = \lambda / U \quad (2.1)$$

where L is the insulation layer thickness (m) and λ is the material-specific thermal conductivity ($W \cdot m^{-1} \cdot K^{-1}$), and also using the Equation (2.2):

$$m = \varrho \times L \times A \quad (2.2)$$

where ϱ is the raw density ($kg \cdot m^{-3}$), the necessary mass m (kg) of each square meter of functionally equivalent insulation material was derived. This approach is equivalent to the one used by Schmidt et al. [27]. This FU choice allows us to cope with the multi-functionality of the systems and allows us to compare bio- and fossil-based products. By relating the systems to the same output, the criteria become comparable for the two resource usage strategies.

- *Criteria for Assessment*

For the FU, we calculated the environmental impacts of both strategies over the complete life cycles of the insulations. We used the two resource-use criteria *cropland* and *fossil fuel demand*, which are life cycle inventory indicators, and the emission-based criterion *GHG emissions*, which is a life cycle impact assessment indicator. We did not further weight or aggregate the three criteria, like how it is done in multi-criteria approaches, such as for example [28]. Instead, we treated them as equally important.

We calculated the direct *cropland demand* to grow the biomass as well as the cropland required within background processes as $m^2 \cdot FU^{-1}$. The data was taken from ecoinvent and others (see Tables 5.2.1-5.2.4) from the land use category "arable land" (CORINE definition level 21, [29]).

We aggregated the *fossil fuel demand* to MJ per FU for both strategies using lower heating values (crude oil: 43.2 $MJ \cdot kg^{-1}$, natural gas: 47.3 $MJ \cdot kg^{-1}$) [30,31]. We took the data for crude oil and natural gas demands from life cycle inventories in the respective datasets (Tables 5.2.1-4). This choice of fossil fuels resources came from the composition of the fossil-based insulation (see sub-section in 6.2.2 *EPS Production…*).

Finally, we aggregated *GHG emissions* contributing to climate change, such as carbon dioxide (CO_2), methane and nitrous oxide, to CO_2 equivalents per FU (kg $CO_{2e} \cdot FU^{-1}$). We used characterization factors for a time horizon of 100 years from IPCC (CO_2: 1, methane: 25, nitrous oxide: 298) [32].

Biomaterial Strategy

- *European Hemp Market Conditions*

Hemp (*Cannabis sativa* L.) has been used in Europe as an industrial crop for many years. The majority was cultivated in France in stable amounts (EU–27: 10,600 ha in 2010) [33,34]. For the time being, no large-scale structural effects on the hemp market were expected, which is a postulate for using the substitution approach for co-product accounting in LCA: Enough production capacities for hemp processing exist to meet a growing demand [33]. Furthermore, markets already exist for the co-products hemp seed as birdfeed and shives as animal bedding, as well as the respective alternative products in these sectors.

- *Hemp Cultivation and Processing to Insulation*

In the *biomaterial* strategy, we considered the supply chain from the cultivation of hemp, to the provision of fossil fuels, to transport processes up to the provision of heat and electricity, and finally to the end-of-life stage. We took the data for fiber hemp cultivation representing European conditions from [35], and others (Table 6.8).

Table 6.8: Characteristics of hemp (*Cannabis sativa* L.) cultivation and processing (*biomaterial* strategy)

	Unit	Amount	References
Nitrogen ∣ phosphate ∣ potassium fertilizer ∣ hemp seeds	$kg \cdot ha^{-1}$	80 ∣ 100 ∣ 180 ∣ 45	[35]
Straw (15 % water content) ∣ Fiber yield (technical fibers 22 %)	$Mg \cdot ha^{-1}$	8.00 ∣ 1.76	[35]
Co-product hemp seeds	% *w*/*w* of total yield	10	[36]
Co-product hemp shives	% *w*/*w* of straw yield	57	[33]
Tractor employment (all necessary activities)	$h \cdot ha^{-1}$	6	[37]
Emissions from and resource demand of the production of Polyester fiber ∣ Sodium hydroxide	$kg\ CO_{2e} \cdot kg^{-1}$ $m^2 \cdot kg^{-1}$ $MJ \cdot kg^{-1}$	4.43 ∣ 1.43 8.3×10^{-5} ∣ $3.8 \cdot \times 10^{-5}$ 79.42 ∣ 10.22	based on Eco-profiles of EU plastics industry [38]
Transports			
• Field to processing ∣ processing to insulation production	km	150 ∣ 235	[36]
• Production to distributor ∣ distributor to building	km	400 ∣ 40	analogue to EPS
• Building to end-of-life	km	40	analogue to EPS

[a] EPS—expanded polystyrene

After harvesting, the hemp bales were processed to long fibers and co-products. The long fibers were finally bonded to insulation mats, consisting of a mixture of hemp (83 %) and polyester fibers (12 %), and were impregnated with sodium hydroxide (5 %) as a flame retardant [36]. We assumed that the necessary electricity for insulation processing was generated from fossil fuels in an oil-fired and a gas-fired cogeneration plant. We used product-specific data for a hemp mat, for which a CO_2 balance study was available [36].

Following Equations (2.1) and (2.2), 1 m^2 of the insulation material hemp mat with $U = 0.2$ $W \cdot m^{-2} \cdot K^{-1}$ and a thermal conductivity $\lambda = 0.04$ $W \cdot m^{-1} \cdot K^{-1}$ weighed 6 kg and had a layer thickness of $L = 0.20$ m.

- *Land Use Change Effects*

Land use change (LUC) effects can be crucial for the climate change impacts of bioresources, such as bioenergy crops [39] or bioplastics [40]. LUC can also affect biodiversity or have other environmental impacts [41]. However, the latter were out of the scope of this study.

We did not expect direct LUC (dLUC) for hemp cultivation: Management practices are similar to other annual crops. Accordingly, relevant changes in carbon stocks in above- and/or below-ground biomass as well as in soil-bound carbon are unlikely [42].

Hemp competes with cash crops like cereals, oleiferous plants, or bioenergy crops for agricultural land. It is usually considered to be a crop with positive effects within rotation systems, i.e., subsequent crops need no or less herbicides, whereas hemp itself needs none. Thus, unproductive sites can be used [35]. Provided that fiber demand grows (e.g., due to additional insulating efforts for buildings), we presumed that hemp would be integrated into existing crop rotation systems. For the calculation of indirect land use change (iLUC) effects on GHG emissions resulting from land demand for crops, global iLUC factors were published, for example 1.43 Mg CO_{2e}·ha^{-1} [43] or 5 Mg CO_{2e}·ha^{-1} [44]. As a conservative value, we applied the former factor to the gross value of land cultivated with hemp in a separate examination. We did not assign such factors to the system analyses in general because such iLUC is still marked by high uncertainties [45–47].

- *Co-Products*

Hemp cultivation yields seeds and shives as market-relevant co-products. Hemp seeds are mainly used for animal feed (70 %), especially in birdseed (4,000 metric tons in 2010 in Europe) [33]. Sunflower (*Helianthus annuus* L.) seeds could be their alternative in feed compositions, even though they do not match hemp seed's nutrient composition completely [48]. We assumed integrated Swiss farming systems as sunflower producers with yield levels around 3 Mg·ha^{-1} [49]. Hemp shives are mainly used for animal bedding [33], where alternatively wheat (*Triticum aestivum* L.) straw can be used. The third co-product was excess heat from the electricity co-generation process. Following the substitution approach, we determined credits for avoided GHG, energy and cropland demand according to those emissions and demands which would result from the alternative products and processes (Table 6.9).

Table 6.9: Co-products in the *biomaterial* strategies and their alternatives (A), and credits for end-of-life energy recovery

Co-product ǀ Alternative A	Unit [a]	Amount	Remarks [b]	Ref.
Hemp Seed	kg·FU^{-1}	2.55	Substitution rate 100 %; substituted by sunflower seeds in birds" feed	[48]
A: Sunflower Seed	kg CO_{2e}·kg^{-1} m^2·kg^{-1} MJ·kg^{-1}	1.02 ǀ 1.24 3.6 ǀ 12.1 4.835 ǀ 4.928	Production conditions: integrated Swiss ǀ conventional Spanish #235 ǀ #6961	[49]
Hemp Shives	kg·FU^{-1}	9.00	Substitution rate 62 %; by wheat straw in animal bedding	[33]
A: Wheat straw	kg CO_{2e}·kg^{-1} m^2·kg^{-1} MJ·kg^{-1}	0.08 0.2 0.519	Integrated Swiss production conditions; inputs and outputs of wheat cultivation are economically (7.5 %) allocated to the straw (#240))	[49]
Co-generated heat	MJ·FU^{-1}	22.7	Excess heat from electricity generation (from fossil fuels)	[38]
A: Heat at industrial furnace from fuel oil ǀ natural gas	kg CO_{2e}·MJ^{-1} m^2·MJ^{-1} MJ·MJ^{-1}	0.09 ǀ 0.08 $3.4 \cdot 10^{-7}$ ǀ $5.7 \cdot 10^{-8}$ 1.287 ǀ 1.285	#1582 ǀ #1352	[49]
Credits for energy-recovery (Crude oil ǀ natural gas)	MJ·kg^{-1}	0.24 ǀ 18.96	From waste co-incineration after detaching from building; total credit is 23.7 MJ·kg^{-1} according to dataset 2.22.01; 1% is from crude oil, 80 % from natural gas	[11]

[a] m^2 taken from land use category "arable land" in the ecoinvent datasets, representing CORINE definition level 21 [29]; [b] # indicates reference number of respective ecoinvent datasets [49]

For the end-of-life stage, we derived fossil fuel and GHG emission credits from environmental product declarations: For the co-incineration of detached insulations, the reference states a credit of non-renewable primary energy of 23.7 MJ·kg^{-1} insulation, of which 80 % are from natural gas and 1 %

is from crude oil [11]. This source calculated the values based on the assumption that the German electricity supply mix and a heat generation from natural gas are displaced by the co-incinerated insulations. The resulting total fossil fuel credit was 19.2 MJ primary energy per kilogram of insulation.

BBSR datasets already offset the CO_2 credits for biogenic carbon in hemp against the GHG emissions during waste incineration. Hence, the datasets display only aggregated GHG emissions for the end-of-life stage of hemp and EPS insulations, and we could not display disaggregated biogenic CO_{2e} credits.

Bioenergy Strategy

- *EPS Production and Processing to Insulation*

In the *bioenergy* strategy, we considered the supply chain from the cultivation of the bioenergy crops (optional SRC, or maize), to the provision of fossil fuels, to transport processes, up to the provision of heat and electricity, and finally to the end-of-life stage.

EPS insulations are made from polystyrene granulate, which can be produced from the fossil resources crude oil and natural gas via several pathways. Here, an oil to gas ratio of approx. one to one was assumed [49] (#1835). In the basic variant, the EPS had a 45 % share of recycled material (EPS45). The heat demand of the EPS production process was met by either the gasification of poplar wood chips or by burning biogas from digested maize silage (Table 6.10). We scaled both co-generation processes to supply the heat demand of the EPS production process. The co-generated electricity is not entirely needed for the insulation production process itself (see also sub-section in 6.2.2 *Bioenergy strategy/Co-Products*).

Table 6.10: Characteristics of *bioenergy* co-generation (heat and electricity) from poplar short rotation coppice (*Populus* spp.) via gasification (option SRC) [50,51], and from maize silage (biogas; *Zea mays* L.) (option maize) [52], both for German technology and production characteristics

Co-generation characteristics	**Unit**	***Bioenergy* option**	
		Short rotation Coppice	**Maize Silage**
Yield (wood chips$_{50\%\ wet}$; maize yield$_{dry}$)	$Mg \cdot ha^{-1}$	14.5	14.9
Power plant characteristics:			
Mean transport distance of feedstock	km	30	50
Operating hours of gasification; biogas plant	$h \cdot year^{-1}$	7,000	6,000
Electric efficiency	%	33	33
Installed heat capacity	MW	6.4	1.5
Installed electric capacity	MW	5	0.5

Disposed insulations can be co-fired in waste incineration plants. Our reference states a credit of non-renewable primary energy of 30.2 $MJ \cdot kg^{-1}$ insulation, of which 77 % is from natural gas and 1 % is from crude oil [11]. The source calculated the latter values while assuming a German electricity supply mix and heat generation from natural gas as displaced processes. This corresponded to a total credit of primary energy of 23.56 $MJ \cdot kg^{-1}$ EPS. GHG emissions from co-incineration were presented with 1.2 kg $CO_{2e} \cdot kg^{-1}$.

Following Equations (2.1) and (2.2), 1 m^2 of the insulation material EPS with $U = 0.2\ W \cdot m^{-2} \cdot K^{-1}$ and a thermal conductivity $\lambda = 0.036\ W \cdot m^{-1} \cdot K^{-1}$ weighed 5 kg and had a layer thickness of $L = 0.18$ m.

- *Potential Land Use Change Effects*

The establishment of perennial bioenergy crops like SRC can result in recognizable, positive changes in carbon stocks as a direct LUC effect [53]. However, here we assigned no credits for a potential increase in soil organic carbon to the *bioenergy* strategy. The uncertainty of the amount of carbon sequestration remains still high: Long-term SOC data are missing for sites where SRC plantations were grubbed and re-grown with other crops [54]. We neglected indirect LUC effects of poplar cultivation in Germany [47] because we assumed that SRC replaces low yielding rye cultivation [50]. In an additional examination, we assigned a global iLUC factor [43] to gross SRC cultivation acreage.

LUC effects of maize cultivation as an annual energy crop can be important: In Europe, if maize is cultivated on previous grassland, GHG emissions of 2.6 Mg $CO_{2e}\cdot ha^{-1}\cdot year^{-1}$ are possible [55]. We considered such potential emissions in a scenario analysis (see details for EPS45_LUC below). We generally did not consider indirect effects for maize [47], but again we addressed them with a separate iLUC examination for the gross value of agricultural land planted with maize [43].

- *Co-Products*

In the *bioenergy* strategy, the excess electricity from the co-generation processes for the heat-driven EPS production process was a market-relevant co-product. Accordingly, we credited this strategy with avoiding emissions and resource demand from substituted oil- and gas-fired power plants (Table 6.11). We made this choice of fuels to maintain comparability to the fossil resources for material use. For the time being, we do not expect any large-scale structural effects on the electricity market from the additional electricity feed-in.

Table 6.11: Co-products in the *bioenergy* strategies and their alternatives (A), and credits for end-of-life energy recovery

Co-product ∣ Alternative A	Unit [e]	Amount	Remarks	Ref.
Co-generated electricity • from wood chips gasification • from biogas	$kWh\cdot FU^{-1}$	8.1 ∣ 7.2 7.6 ∣ 6.7	Already reduced by own electricity demand 0.14 $kWh\cdot kg^{-1}$ EPS ∣ 0.3 $kWh\cdot kg^{-1}$ EPS100 #11792 ∣ #11791	[49]
A: Electricity from oil- ∣ gas-fired plant	kg $CO_{2e}\cdot kWh^{-1}$ $m^2\cdot kWh^{-1}$ $MJ\cdot kWh^{-1}$	1.13 ∣ 0.56 $1.9 \cdot 10^{-6}$ ∣ $1.7 \cdot 10^{-3}$ 14.896 ∣ 9.888	#1620 ∣ #1384	[49]
Digestate containing	kg $N\cdot kWh^{-1}{}_{heat}$ kg $P_2O_5\cdot kWh^{-1}{}_{heat}$ kg $K_2O\cdot kWh^{-1}{}_{heat}$	0.003 0.005 0.013	Values already corrected to represent plant-available nutrient content; 30% according to Ref.	[56]
A: CAN [a]-fertilizer	kg $CO_{2e}\cdot kg^{-1}$ N [b] $m^2\cdot kg^{-1}$ N [b] $MJ\cdot kg^{-1}$ N [b]	8.66 $9.1\cdot 10^{-5}$ 51.4	#42	[49]
A: P_2O_5 [c]-fertilizer	kg $O_{2e}\cdot kg^{-1}$ P_2O_5 $m^2\cdot kg^{-1}$ P_2O_5 $MJ\cdot kg^{-1}$ P_2O_5	2.03 $9.2\cdot 10^{-5}$ 18.612	#57	[49]
A: K_2O [d]-fertilizer	kg $CO_{2e}\cdot kg^{-1}$ K_2O $m^2\cdot kg^{-1}$ K_2O $MJ\cdot kg^{-1}$ K_2O	1.44 $6.2\cdot 10^{-5}$ 17.789	#53	[49]
Credits for energy-recovery (Crude oil ∣ natural gas)	$MJ\cdot kg^{-1}$	0.3 ∣ 23.25	From waste co-incineration after detaching from building; total credit is 30.2 $MJ\cdot kg^{-1}$ according to dataset 2.22.06; 1 % is from crude oil, 77 % from natural gas	[11]

[a] calcium ammonium nitrate; [b] nitrogen; [c] phosphate; [d] potassium oxide; [e] m^2 taken from land use category "arable land" in the ecoinvent datasets, representing CORINE definition level 21 [29]; # indicates reference number of respective ecoinvent datasets [49]

Biogas production from maize silage generates the additional co-product digestate. This digestate can be used as a mineral fertilizer substitute in agriculture [56]. The nutrient availability from organic residues can vary very widely between 6 %–80 % [57]. We assumed here that the substitution effect of the digestate was approximately 30 % according to [56]. This is similar to other plant-based organic residues in [57]. We took the data for avoided emissions and resource demand from substituted fertilizers from ecoinvent [49].

Scenario Analyses

For both strategies, we had to make numerous assumptions, for example regarding parameter values or, among others, inclusion/exclusion of processes. We defined the most plausible scenarios to assess the influence of model assumptions and data sources on the strategy comparison (Table 6.12).

In the *biomaterial* strategy, we varied the share of hemp fibers in the insulation material and the production system of sunflower seed. Different mixtures of hemp insulations are available on the market. Therefore, in addition to the basic variant with 83 % hemp fibers and 12 % polyester fibers (Hemp83), we introduced a variant that was made from pure hemp fibers (Hemp95). Both variants were impregnated with sodium hydroxide (5 %) as a flame retardant. For the variant Hemp83_Sunfl, we assumed a different production characteristic for the co-product sunflower seeds when substituting hemp seeds. As several regions could supply sunflower seeds [34], we chose conventionally produced sunflower seeds from Spain instead of seeds from Swiss integrated production. The main difference between regions is their yield and input level, which results in land demand that is three times higher in Spain than in Switzerland.

Table 6.12: Characteristics of the *biomaterial* and *bioenergy* strategies and parameters varied (bold letters) in the different variants of the scenario analyses

	Biomaterial Strategy			*Bioenergy* Strategy		
	Hemp83	**Hemp95**	**Hemp83_Sunfl**	**EPS45**	**EPS_LUC**	**EPS100**
Material Characteristics						
• Hemp share in mat (%)	83	**95**	83	*n/a*	*n/a*	*n/a*
• Soda (%)	5	5	5	*n/a*	*n/a*	*n/a*
• Polyester fiber (%)	12	0	12	*n/a*	*n/a*	*n/a*
• EPS[a] recyclate share (%)	*n/a*	*n/a*	*n/a*	45	45	**100**
• Bulk density ($kg \cdot m^{-3}$)	30	30	30	28	28	28
• Thermal conductivity ($W \cdot m^{-1} \cdot K^{-1}$)	0.04	0.04	0.04	0.036	0.036	0.036
• Weight of 1 m^2 insulation with U=0.2 $W \cdot m^{-2} \cdot K^{-1}$ (kg)	6	6	6	5	5	5
Substitute for co-product; Sunflower seed yield ($Mg \cdot ha^{-1}$)	3.15	3.15	**1.03**	*n/a*	*n/a*	*n/a*
GHG [b] emission factor ($kg \cdot CO_{2e} \cdot kWh_{th}^{-1}$)	*n/a*	*n/a*	*n/a*	0.017 (SRC [c]) 0.253 (maize)	0.017 (SRC [c]) **0.277 (maize)**	0.017 (SRC [c]) 0.253 (maize)
Electricity demand for production process ($kWh_{el} \cdot kg^{-1}$ EPS [a])	*n/a*	*n/a*	*n/a*	0.14	0.14	**0.31**
Description and main effect of variation	Basic assumptions	No additional synthetic fibers in insulation result in higher land demand and reductions in fossil fuels and GHG [b] emissions.	Alternative source for sunflower seed substituting hemp seed from region with lower yield levels (Spain instead of Switzerland)	Basic assumptions	Considering land use change for maize cultivation results in additional GHG [b] emissions.	Insulation production from 100 % recyclate needs more electricity, resulting in higher fossil fuel demand and GHG [b] emissions.

[a] expanded polystyrene; [b] greenhouse gases; [c] short rotation coppice; *n/a* not available

In the *bioenergy* strategy, we varied the share of recycled EPS in the insulation material and the LUC from grassland to maize was included. Synthetic insulation materials may contain different shares of recycled materials. In the basic *bioenergy* variant EPS45, the insulation had a 45 % share of recycled EPS [49] (#11792), whereas in variant EPS100 it was produced entirely from recycled EPS [49] (#11791). The production process of EPS100 had more than twice the electricity demand compared to EPS45. However, the reduction in virgin EPS demand more than compensated for this by reduced CO_{2e} emissions in the EPS pre-chain from 2.6 to 0.6 kg $CO_{2e} \cdot kg^{-1}$ EPS [49,58].

Maize as an energy crop is discussed as a relevant driver of grassland to cropland conversion in Europe. It was identified with 50 % as the dominant land use after grassland conversion in a GIS analysis of four German federal states [59]. Its cultivation in Germany increased to 800,000 ha between 2003 and 2012, whereas grassland decreased by 250,000 ha [60]. Therefore, we assumed that 16 % of maize was cultivated on former grassland, and that associated GHG emissions were 0.277 kg $CO_{2e} \cdot kWh_{th}$ [52]. We considered such LUC emissions in variant EPS45_LUC.

6.2.3 Results and Discussion

Comparison of Strategies and Scenario Analyses

- *Cropland Demand*

We found that *cropland demand* was lower in both of the basic *bioenergy* strategy options than in the basic *biomaterial* strategy (Table 6.13, Figure 6.5). Within the *bioenergy* options, the short rotation coppice needed less cropland than maize silage.

Table 6.13: Net results and deviations (absolute and relative) for the three criteria cropland use, fossil fuel demand and GHG emissions of the two systems (basic assumptions in bold) and their variants in the scenario analyses. Values in parentheses have a global iLUC factor (Audsley 2009) assigned for land demanded for respective biomass cultivation (reproduction of Table 6 in Hansen *et al.* (2016a))

	Cropland Use			Fossil Fuels Demand			Climate Change		
	Net Result	Deviation		Net Result	Deviation		Net Result	Deviation	
	$m^2\ FU^{-1}$	$m^2\ FU^{-1}$	%	$MJ\ FU^{-1}$	$MJ\ FU^{-1}$	%	$kg\ CO_{2e}\ FU^{-1}$	$kg\ CO_{2e}\ FU^{-1}$	%
				Biomaterial strategy					
Hemp83	**21.25**			**8.10**			**10.47 (15.02)**		
Hemp95	24.32	3.07	14	−44.85	−52.95	−654	7.78 (13.00)	−2.69 (−2.03)	−26 (−14)
Hemp83_Sunfl	−0.53	−21.77	−102	7.86	−0.24	−3	9.90 (14.46)	−0.57 (−0.57)	−5 (−4)
				Bioenergy strategy					
				Gasification of wood chips from short rotation coppice					
EPS45	**0.72**			**18.64**			**9.57 (9.68)**		
EPS100	0.72	0	0	−201.86	−220.50	−1183	0.39 (0.49)	−9.18 (−9.18)	−96 (−95)
				Biogas from maize silage					
EPS45	**6.78**			**25.04**			**12.20 (13.17)**		
EPS45_LUC	6.78	0	0	25.04	0.00	0	12.49 (13.46)	0.28 (0.28)	2 (2)
EPS100	6.78	0	0	−195.45	−220.49	−881	3.02 (3.99)	−9.18 (−9.18)	−75 (−70)

Scenario/abbreviation: Hemp95—pure hemp fiber insulation; Hemp83_Sunfl—credited process with lower sunflower yield level; EPS100—pure EPS recyclate; EPS45_LUC—biogas from maize, partly grown on previous grassland

As we expected, the agricultural production processes that provided the bioresources fibers or bioenergy crops dominated this criterion (Figure 6.5). For an equal insulation effect, we found that the *biomaterial* strategy needed three times more land than the maize *bioenergy* option or approximately thirty times more than the SRC *bioenergy* option. The gross land demand in the *biomaterial* strategy was 31.84 $m^2 \cdot FU^{-1}$, with an additional 4.60 $m^2 \cdot FU^{-1}$ being necessary if the insulation material was produced from pure hemp (variant Hemp95).

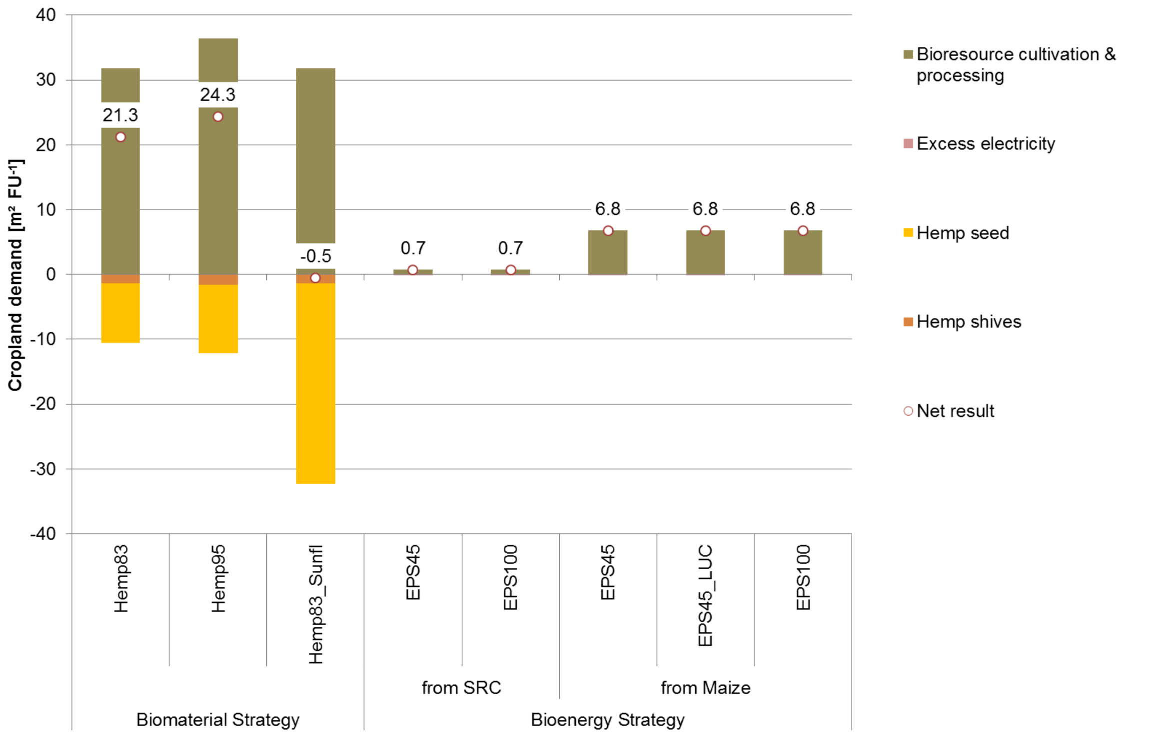

Figure 6.5: Resource demand—Cropland—from system processes in the *biomaterial* and *bioenergy* strategies as well as credits from co-products; numbers indicate net result for each strategy and its variants from scenario analyses (see descriptions in Table 6.12)

We only obtained significant land credits for avoided cultivation of sunflower seeds, which we substituted for the co-product hemp seed in the *biomaterial* strategy. Our substitution choice had a relevant influence on the outcome of the strategy analysis: If we selected a less productive system as a reference (Hemp83_Sunfl), the calculated net land demand for producing a hemp mat as insulation material could even be negative (-0.52 $m^2 \cdot FU^{-1}$). Mean European yield levels for sunflower in 2014 were 1.44 $Mg \cdot ha^{-1}$ [61]. Hence, the Swiss seeds that we assumed here in the basic variant represented a very productive sunflower system with its yields of 3.15 $Mg \cdot ha^{-1}$, whereas the Spanish seeds represented the least productive ones (1.03 $Mg \cdot ha^{-1}$). The basic variant Hemp83 could accordingly be seen as a worst-case scenario and the *biomaterial* strategy becomes the preferable one with the caveat that superseded sunflower systems have low yield levels.

However, negative results must be interpreted with care. An oft-used interpretation for negative criteria values resulting from the substitutional approach is that the respective resource, in this case the cropland, is set free ("The overall impact of the system is more than compensated by the avoided impact the co-functions have elsewhere" [20], (p. 78). In this study, the characteristics of the "set free" land, for example its soil fertility and climatic region and its respective farming specifications, are suitable for low-yielding sunflower cultivation. However, in general, it is not possible to infer its suitability for other agricultural purposes.

In the *bioenergy* strategy, we obtained hardly any cropland credits for excess electricity. We found that the variation in material characteristics, i.e., the share of recyclate, did not influence the cropland demand because the same amount of energy crops was needed, as the production of both virgin and recycled EPS insulations needed the same amount of thermal energy. Even though the electricity demand of EPS100 production is higher than that of EPS45 (see sub-section in 6.2.2 *Scenario Analyses*), the total amount of the by-product "co-generated electricity" still exceeded this demand. Accordingly, no additional cropland was needed for energy generation.

- *Fossil Fuels Demand*

With regard to *fossil fuel resource demand*, the *biomaterial* strategy seemed to us to be preferable to the *bioenergy* strategy (Table 6.13, Figure 6.6).

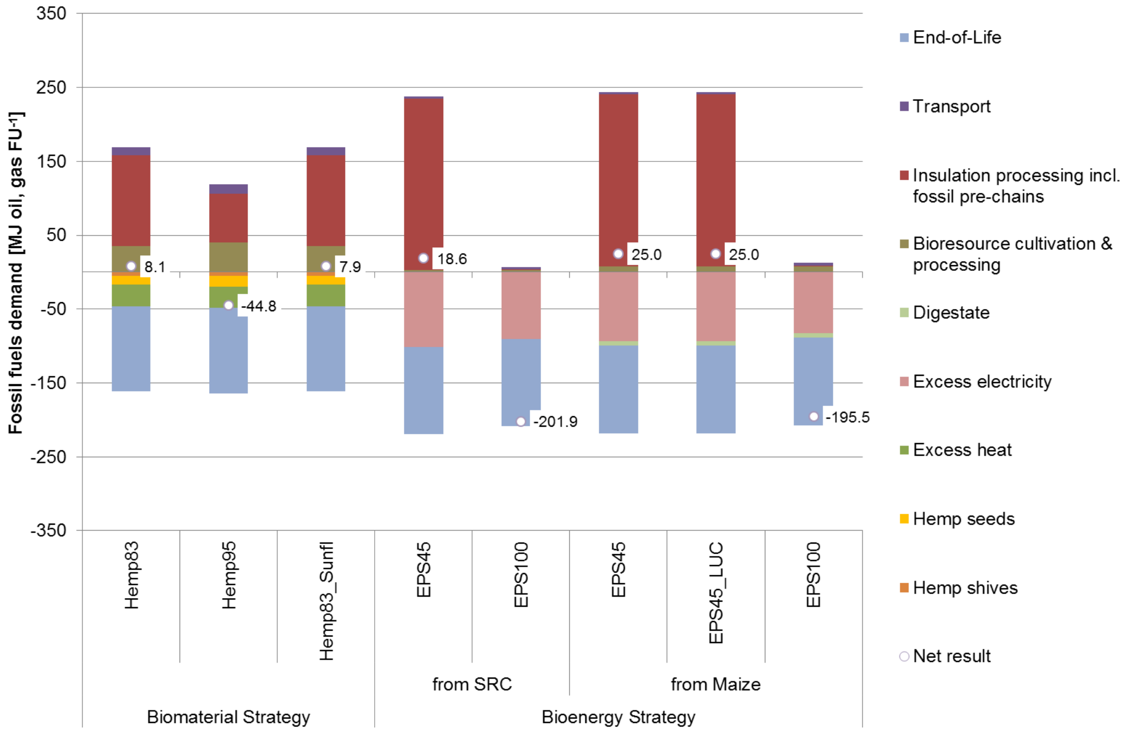

Figure 6.6: Resource demand —Fossil fuels (crude oil, natural gas)—from system processes in the *biomaterial* and *bioenergy* strategies as well as credits from co-products; numbers indicate net result for each strategy and its variants from scenario analyses (see descriptions in Table 6.12)

The insulation production process dominated the *fossil fuel demand* in both of our strategies, with demands ranging between 66.07 and 232.35 MJ·FU^{-1}, except for the *bioenergy* variant with pure recycling material (EPS100; 1.11 MJ·FU^{-1}).

The end-of-life phase obtained relevant credits due to co-incineration in both of our strategies. In the *bioenergy* strategy, we substituted excess electricity for fossil resource-intense processes; this contributed as many credits as the end-of-life stage. We found that the substitution of mineral fertilizers by the co-product digestate had only a marginal effect, with 3 % of total fossil fuels credits. In the *biomaterial* strategy, we found that the co-generated heat provided relevant credits, whereas the other co-products seed and shives added only minor ones. As with the cropland demand, our substitutional LCA approach resulted in a negative net fossil fuel demand in some variants.

Our basic *biomaterial* strategy needed nearly 30 % of the net amount of fossil fuels compared to maize or 40 % compared to SRC *bioenergy*. From the ecoinvent data for pure recyclate (EPS100), we determined considerable negative net fuel demands of approximately −200 MJ·FU^{-1} for the *bioenergy* strategy (−201.86 MJ·FU^{-1} for SRC, −195.45 MJ·FU^{-1} for maize). Using that data set would change the rating of the strategies and would make the *bioenergy* strategy preferable. The reason was how recycling material was defined in these datasets: Still-unused insulations that are collected from municipalities and builder's merchants, consisting of clean, sorted offcuts, were re-processed to a so-called 100 % recycling material. The previous resource demand from original material production was not considered. The dataset implied recycling, even though the first life cycle was not completely closed, as this material was not attached to a building as insulation. Furthermore, assuming best practice on the construction sites, only a small fraction of such virgin-like EPS would be available for the transformation into the recycling material. We think this balancing approach is debatable. Currently, we believe that recovery of EPS from dismantled insulation material after its service life is far from a practical implementation due to contaminants and adhesives, and hence we chose co-incineration as the end-of-life option.

Under the assumption that such a recycling becomes possible in the future, we believe that the fuel demand for the recyclate share must be accounted for as well. We estimated the fuel demand for a pure virgin material (EPS0) from the EPS45 dataset for the *bioenergy* SRC option by allocating its entire energy demand of the production process to the 55 % share of virgin EPS (232.35/0.55) (Table A1). The resulting increased net fossil fuel demand of 209 MJ·FU^{-1} indicates the importance of recycling for this strategy. Its net fossil fuel demand is one order higher than that of the *biomaterial* strategy, no matter if recycling is considered there. The *bioenergy* variant EPS100 yielded net negative fossil fuel demands of −2.5 MJ·FU^{-1} from the second use cycle onwards. For the same benefit after three use cycles, we found that a minimum of 75 % recyclate would be necessary (Table A1).

For bioresource-based insulations in the *biomaterial* strategy, explicit data on hemp recycling are scarcely available and thus, we did not assess recycling here.

From *fossil fuel demand*, we conclude that recycling concepts for materials are far more important in terms of fossil resource conservation than the choice between fossil or biogenic resources as raw materials. A comparable conclusion was reached by Colwill et al. [62], who claimed an efficient material use through effective end-of-life management and good design.

- *Contribution to Climate Change*

We found that the net contribution to climate change was in the same range for the biomaterial and the SRC bioenergy strategy, whereas the bioenergy option from maize (EPS45_maize) emitted more GHG (Table 6.13, Figure 6.7). An increase in the amount of recyclate in the EPS insulation (EPS100) reduced emissions in a way that the bioenergy strategy would be favorable instead. Accounting for iLUC effects increased the difference between the strategies in the same way (Table 6.13).

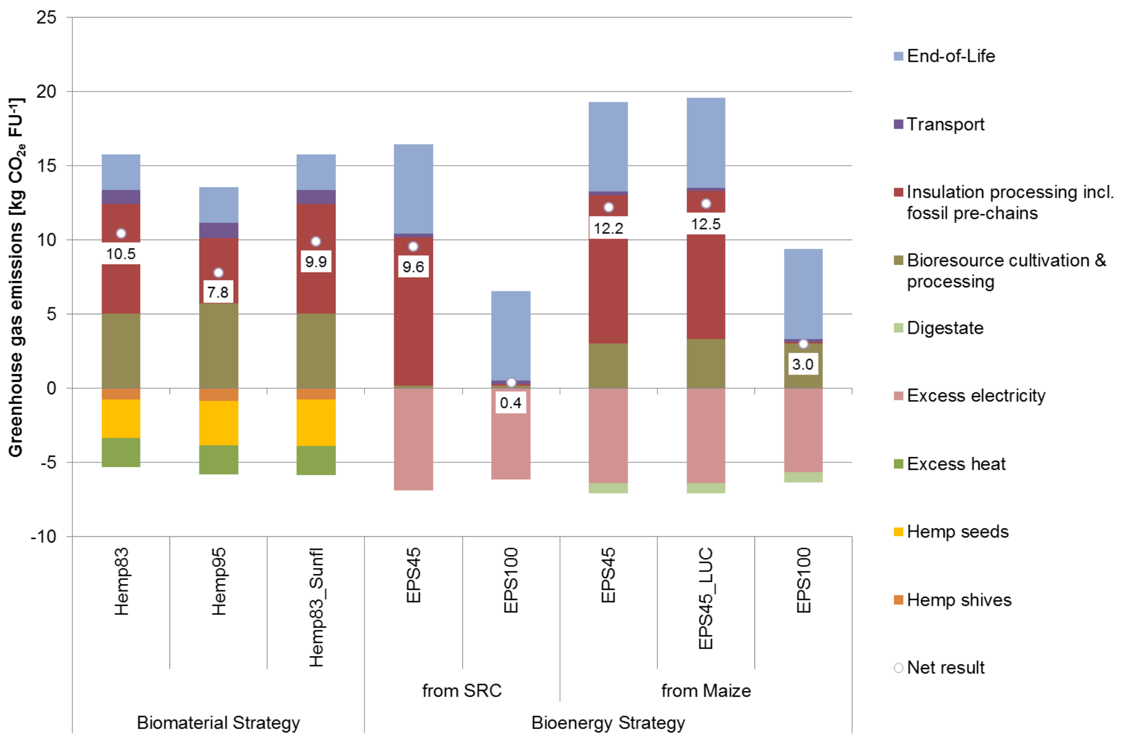

Figure 6.7: GHG emissions from system processes in the *biomaterial* and *bioenergy* strategies as well as credits from co-products; numbers indicate net result for each strategy and its variants from scenario analyses (see descriptions in Table 6.12)

In both of our strategies, the insulation processing contributed the most to climate change. In the biomaterial strategy, hemp cultivation contributed the second-most GHG emissions, whereas in the bioenergy strategy the emissions from the end-of-life incineration of detached insulation had the second highest share. We found that such end-of-life emissions were of lesser importance in the biomaterial strategy (2.38 kg CO_{2e}·FU^{-1}), because mostly biogenic carbon was re-emitted.

We found that the use of the co-product digestate in the bioenergy maize option contributed 10 % of the GHG credits. The co-product excess electricity in the bioenergy strategy substituted electricity from carbon-intensive electricity production from fossil fuels, thus yielding far more credits.

By changing the material composition by an increase in the hemp fiber share in the biomaterial strategy (Hemp95), we would notably reduce GHG emissions by 26 %. Another option to reduce the GHG emissions from the biomaterial strategy is the substitution of polyester fibers with fibers from maize starch. Such pre-existing technologies were analyzed in a CO_2 balance for two hemp insulation materials [36]. Nevertheless, we found that pure hemp insulation without any additional fiber materials would still have a lower GHG impact.

The *bioenergy* EPS100 variants had a generally low climate change contribution (≤3.02 kg CO_{2e}·FU^{-1}).

If we included the emissions from direct land use change in the *bioenergy* strategy (EPS45_LUC), the net contribution to climate change amounted to 12.5 kg CO_{2e}·FU^{-1}, which was only 2 % higher than that of the basic EPS45. When we accounted for indirect LUC effects on GHG emissions by a global emission factor, which is applicable to the land demand for the respective agricultural commodities hemp, SRC, or maize, this increased the difference between both strategies. The *biomaterial* strategy then had nearly 50 % (compared to SRC) or 15 % (compared to maize) higher emissions than the *bioenergy* strategy (Table 6.13). When we applied a recently published, less conservative global iLUC factor [44], this even resulted in 165 % or 70 % higher emissions.

Discussion of Market Implications and Reference Systems

It is still unclear whether high recycling quotas for insulations are achievable on a larger scale. The available data in this study for the product from recycled EPS stemmed from a single company acting in a limited region, so this variant could be interpreted as a best case scenario. Furthermore, we are not aware of technologies that would make it possible to detach old EPS insulation from buildings economically. More information on several end-of-life options is necessary to identify the best choice for the end of life of building materials [63], which was not available to us. Comprehensive information becomes even more important if additional environmental impacts like acidification or eutrophication are to be assessed for such bio-based materials [64].

The complexity of the involved systems made the substitutional approach challenging. Markets do exist for the hemp co-products seeds and shives. However, if the market placement is misjudged, possible side or rebound effects of the multi-product systems might be overlooked.

Under the geographic conditions analyzed in this study, we classified hemp seeds as valuable bird feed that cannot easily be substituted. Globally, other markets for hemp seed also exist: In Canada for example, hemp seed is mainly grown for human nutrition [65]. Accordingly, different alternative co-production processes would need to be identified if the assessment is done for different geographical regions. We conclude that bioresource usage should always be assessed with a regional focus.

For the second co-product shives, straw was assumed as an alternative bedding material. The market effects of straw use are difficult to anticipate. This is because this agricultural residue is a bioresource which can alternatively be used for bioenergy generation; this has already been implemented, for example in Denmark (approx. 5 TWh in 2012) [66]. If more straw is used as energy feedstock, there might be an even higher demand for straw substitutes on the animal bedding market.

Alternatively, hemp shives can be burned instantly instead of being credited with straw. The calorific values of hemp shives and straw can be assumed to be similar (18.8 vs. 18.7 MJ·kg^{-1} dry matter) [67]. Their incineration would accordingly yield similar energy credits with regard to mass, but land demand of the *biomaterial* strategy might be different.

Ongoing research is examining further marketable options for hemp fiber co-products. For instance, technologies are being proposed to increase the marketable share of shives and short fibers for technical applications [68]. This would increase the credits obtained for shives by the strategy.

However, from the information available at present, we considered that the assumption that straw substitutes for hemp shives in animal bedding, is the most appropriate one.

Nevertheless, we believe our discussion above clearly indicates how the complexity of the involved bioresource systems makes the substitutional approach challenging.

Excluded Effects

In different climatic regions, an identical insulation might yield lower CO_{2e} benefits depending on the initial thermal insulation state of the building [69]. Thermal comfort ranges of inhabitants as well can have an important impact [70]. We neglected this due to our assumption that the buildings have the same initial thermal insulation state and are situated in the same climate. As already stated in Section 2.1.3, we excluded heating and cooling demand from the analysis because its equality in both systems is the functional unit that allows comparing both usage strategies.

The excluded scaling effects concern the assumed parity between material and energy use of bioresources. At higher emission taxes, bioelectricity was found to be more beneficial in terms of absolute emission reduction than the use of biomass as feedstock for bulk chemical production [64,71]. This was because of the large size of the electricity sector and, on the other hand, higher leakage effects for the non-energy use of biomass. We did not implement such cross-sector effects here.

To allow scaled up and geographically broader assessments in the future, we believe it is necessary to implement regional aspects (for example policy instruments—incentives, regulations, sanctions—[72], yield levels, supply chain networks, waste handling) and to adapt assumptions to market developments in order to consider leakage effects adequately. Additionally, a dynamic approach could improve how sequestration effects are represented in the bio- as well as the fossil materials [73]. This includes temporary carbon storage in biomaterials as suggested by Jørgensen et al. [74].

Our study focused on environmental impacts and did not analyze social and economic effects. In the future, comprehensive data might be available for social sustainability assessment, for example from the Social Hotspots Database (SHDB) [75], to facilitate data collection even in complex systems as those under study here. For an even more complete picture of the sustainable resource use of land and fossil fuels, also explicit economic assessments were to be wished for. At the moment, economic impacts are only implicitly considered, for example in the discussion of reference systems or iLUC effects.

There are other aspects that are difficult to assess: bioresource cultivation can be associated with changes in the agricultural landscape or agricultural management practices with complex effects on, amongst others, soil quality, diversity of agriculture [76], or economic opportunities for the farmers [77]. We think that such aspects are not reliably detectable with the criteria *cropland* and *fuel demand* and *GHG emissions* used here, and hence we did not take them into account. An ongoing discussion concerns the indirect effects if agricultural commodities are produced. That is, if cropland is grown with another crop and the previously produced commodity is supposed to be grown somewhere else, triggering changes in land occupation and/or production intensities as well in above- and below-ground biomass stocks. Even though uncertainties are still remarkable with regard to the size of land affected, first approaches are available for accounting for the GHG-related effects. These apply additional GHG emission factors per land used [43,44]. However, we are not aware of such factors for criteria other than GHG, for instance resource demand, which was a main criterion in this study.

6.2.4 Conclusions

Under the assumptions we made, none of the resource usage strategies operated best in all three categories. With a clear focus on sustainable land use, we conclude that a *bioenergy* strategy would be preferable, whereas a *biomaterial* strategy would maintain fossil resources. However, scenario analyses indicated to us that improved recycling concepts could make the *bioenergy* SRC strategy the preferable one in all criteria. Hence, this strategy could be seen as a promising way to exploit additional mitigation potentials in the building sector besides the direct reduction effects of insulations.

For complex production systems, such as the ones exemplarily analyzed in this study, the substitutional assessment approach is challenging and consequently, we strongly advise a combination of criteria for supporting strategy decisions in order to help reduce the impact of uncertainties. In the future, this could be supported by multi-criteria decision making-approaches. Furthermore, we conclude that bioresource systems should be analyzed with a clear regional focus.

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Author Contributions

The individual contribution and responsibilities of the authors are as follows: Jörn Budde and Anja Hansen designed the research. Anja Hansen defined the systems, modeled and analyzed the data and mainly wrote the manuscript. Annette Prochnow contributed to structuring the study and revising the manuscript. All authors have read and approved the final manuscript.

Conflicts of Interest

The authors declare no conflict of interest. The funding sponsors had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, and in the decision to publish the results.

Abbreviations

The following abbreviations are used in this manuscript:

BBSR: Bundesamt für Bau-, Stadt- und Raumforschung im Bundesamt für Bauwesen und Raumordnung
CAN: calcium ammonium nitrate
CO_2: carbon dioxide
CO_{2e}: CO_2 equivalents
dLUC: direct land use change
EPS: expanded polystyrene
EPS45: EPS with a share of 45 % recyclate
EPS45_LUC: EPS with a share of 45 % recyclate and land use change effects considered
EPS100: pure recyclate EPS
EU: European Union
FU: functional unit
GHG: greenhouse gas(es)
Hemp83: hemp insulation containing 83 % hemp fibers
Hemp95: hemp insulation containing 95 % hemp fibers
iLUC: indirect land use change
K_2O: potassium oxide
LCA: life cycle assessment
LCC: life cycle costing
LUC: land use change
MJ: megajoule

N: nitrogen
P_2O_5: phosphate
SLCA: social life cycle assessment
SRC: short rotation coppice

Appendix A

Table A1: Approximated net fossil resource demands (d; MJ·FU^{-1}) of the *bioenergy* strategy (wood chip gasification) for a range of material compositions of recycled and new expanded polystyrene and different use cycles of the recycled material. *Italic* cells indicate material compositions that would result in net resource credits. Values are calculated as proxies from EPS45 dataset (#11792; ecoinvent Centre, 2010) according to d = −213.7 + (%_new/100 + (100-%_new)/100 × 1/use_cycle) × 422.4. Basic variant EPS45 displayed in bold letters

Share of Expanded Polystyrene in the Insulation Material		Use Cycles				
%_recycled	**%_new**	**1**	**2**	**3**	**4**	**5**
100	0	208.8	*−2.5*	*−72.9*	*−108.1*	*−129.2*
95	5	208.8	8.1	*−58.8*	*−92.2*	*−112.3*
90	10	208.8	18.7	*−44.7*	*−76.4*	*−95.4*
85	15	208.8	29.2	*−30.6*	*−60.6*	*−78.5*
80	20	208.8	39.8	*−16.6*	*−44.7*	*−61.6*
75	25	208.8	50.3	*−2.5*	*−28.9*	*−44.7*
70	30	208.8	60.9	11.6	*−13.0*	*−27.8*
65	35	208.8	71.5	25.7	2.8	*−10.9*
60	40	208.8	82.0	39.8	18.7	6.0
55	45	208.8	92.6	53.9	34.5	22.9
50	50	208.8	103.1	67.9	50.3	39.8
EPS45	**55**	**208.8**	**113.7**	**82.0**	**66.2**	**56.7**
40	60	208.8	124.3	96.1	82.0	73.6
35	65	208.8	134.8	110.2	97.9	90.5
30	70	208.8	145.4	124.3	113.7	107.4
25	75	208.8	156.0	138.4	129.5	124.3
20	80	208.8	166.5	152.4	145.4	141.2
15	85	208.8	177.1	166.5	161.2	158.1
10	90	208.8	187.6	180.6	177.1	175.0
5	95	208.8	198.2	194.7	192.9	191.9
0	100	208.8	*n/a*	*n/a*	*n/a*	*n/a*

n/a – not available

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6.3 Proposal for an Evaluation Criterion for Sustainable, Resource-Efficient Biomass Use

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6.3.1 Introduction

Carbon (C) is an essential part of life on earth; approximately 50% of dry plant biomass is carbon (Table 6.14). All organisms rely on C in their metabolism, for example, to generate body tissue and energy carriers [1].

Table 6.14: Carbon content of different organisms (% of dry matter)

Organism	Mean carbon content [% of dry matter]	Range	Ref.
Overall mean of energy crops	46.5		[2]
Maize (*Zea mays* L.) (whole plant)	48.6	47-50.2	[2]
Poplar (*Populus* spec.) (in wood)	47.5		[2]
Willow (*Salix* spec.) (in wood)	47.1		[2]
Wheat (*Triticum* L.) (whole plant)	45.2		[2]
Rye (*Secale cereale* L.) (whole plant)	48.0		[2]
Grasses	43.9	41.4-46.4	[2]
Hemp (*Cannabis L.*)	45.12		[3]
Bacteria	≈ 50		[4]
Microalgae (green/brown)	54.4/24.7	49-58/24-25	[5]

In the same way, human society relies on carbon, particularly in the form of chemical compounds:

- as carbohydrates and fats in food & feed,
- as hydrocarbons in energy carriers and
- as bulk chemicals for the chemical industry.

Humans are dependent on C as a production factor, whether derived from biogenic or fossil sources. However, this perception is not widely held: C is mostly addressed in the context of climate change in its bonded form in greenhouse gases (carbon dioxide: CO_2, methane: CH_4) or as a pollutant (e.g., in volatile organic compounds). Hence, it is more common to see C as a threat instead of an indispensable resource. The interest in its climate change impact is reasonable because the C buffer capability of the atmosphere and other natural sinks is limited [6].

The perception of C in CO_2 as a production factor has progressively gained interest, which can be observed from conference topics [7], from the increasing number of studies on so-called 'dream reactions' [8], by which CO_2 is (re-)transformed into organic compounds as chemical bulk material [9], as well as from a journal with special focus on this topic since 2013 [10]. This interest is demonstrated by a three-digit increase in low-carbon studies since 2011 [11], representing societies' approval of a transition from a fossil-based economy to a bio-economy ('low-carbon'). As stated above, one main reason for this transformation is to avoid climate change by using recently fixed rather than fossilized carbon compounds. However, sustainability assessments of biomass production and usage in the context of a transition to a bio-economy should include more than just their climate impact, and reliable indicators are needed [12,13]. Impact-oriented assessments have to address several methodological caveats, which are discussed in depth in scientific literature on climate impact. Among them are the appropriate choice of reference systems [14], the assessment of indirect effects [15], the allocation in multi-product systems [16], the CO_2 neutrality assumption for biomass [17], and changes in metrics (e.g., global warming potentials change with the new release of IPCC reports [18]). Such restrictions are also relevant for other impact assessments.

Following a paradigm shift from mainly impact-oriented perception (for example, climate change) to a productivity-oriented one, we suggest consideration of C in biomass as a limited resource. Although C is abundant in its gaseous form as CO_2 in the atmosphere, its transformation into biomass C is a demanding process, for instance needing energy, land, water, and nutrients. Furthermore, C is a resource that cannot be substituted by other elements, and the strong sustainability concept needs to be applied [19]. We can assess its appropriate use with the methodological concept of 'productivity', which is common in economics. Productivity is an indicator for the use of limited resources and is expressed as an output/input ratio, e.g., Hill [20]. Several published approaches use the productivity concept to assess C, for instance, technology- [21], sector- [22] or country-specific [23,24] (for more approaches, please refer to Appendix Table A1)[10]. However, their common focus is the cost-efficient reduction of CO_2 emissions to avoid climate change [21]. Recently, the limitation of resources has led to approaches that seek to decouple economic growth and resource use by switching from linear to more circular economic models. Circularity indicators were presented to measure their success [25]. However, they focus on non-renewable resources and have only limited applicability for renewable materials, such as biomass.

In this manuscript, we adopted the productivity concept for a new indicator, Carbon Utilization Degree (CUDe), and apply it to two case studies. CUDe aims to assess efficient C use in production chains. Our objectives are:

- to extend the perception of C (and accordingly CO_2) from having a negative impact to being an indispensable, limited resource and
- to provide a supplementary indicator for policy decision support to express efficient C use in production chains.

6.3.2 The *Carbon Utilization Degree* Approach

Productivity is generally defined as the ratio of output to input (r=output/input), sometimes expressed as a percentage (r_p=output/input*100) [20]. Accordingly, productivity increases if more output is produced from the same input. This can occur if losses are reduced or if additional outputs are generated, for instance, by putting waste to use.

We adopt this concept and propose *CUDe*[11] as a supplementary indicator for policy decision support to assess biomass conversion technology. It expresses the productive carbon fraction of the biomass that is utilized in biomass conversion chains. CUDe is defined as the ratio of finally productive carbon to the carbon that was originally available in the biomass. Carbon is considered productive in an anthropocentric view if it provides a useful output, i.e. it:

- is transformed into marketable products or provides useful services, e.g., insulation material, forage or energy generation (direct benefit)
- performs important ecological functions, e.g., improves soil fertility (indirect benefit).

The approach to calculate the *Carbon Utilization Degree* for a biomass conversion pathway follows a process chain assessment and comprises five steps plus an analysis step (Figure 6.8).

[10] Available as Table 4.6 in this document

[11] We use the term '*CUDe*' to avoid confusion with other concepts: '*C productivity*', which was defined as the specific GDP/CO_2 in Kaya & Yokobori [23], or with '*C efficiency*', which was defined as the ratio of the target level of C emissions and the actual level of C emissions of an economy in Yang [22]. *Productiveness*, as another possible term, has a different meaning than productivity in an economic context ("[…]productiveness (or productive capacity) is a measure of the quality of being productive or having the capacity to produce."[26]).

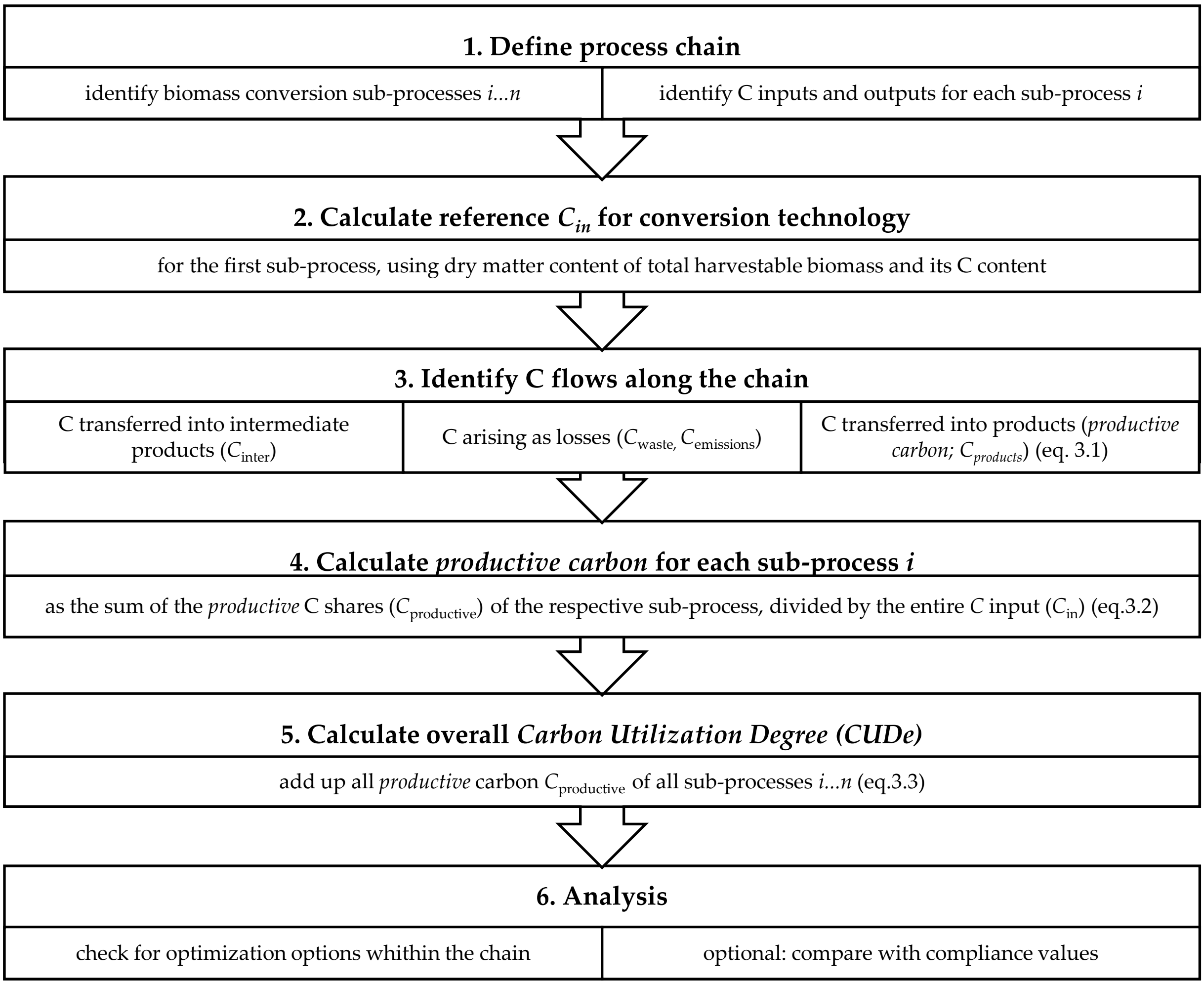

Figure 6.8: Workflow to calculate the Carbon Utilization Degree (*CUDe*) of biomass conversion technologies plus an analysis step. For details, please refer to equations (3.1)–(3.3)

First, all relevant transformation processes in the chain need to be identified and described. In the second step, the system's carbon input is defined as reference value: The amount of fixed carbon C_{in} in the harvestable biomass that enters the process chain is calculated. Third, for each of the subsequent processes, the C balance is determined. C is considered in the following flows: transfer to the chain boundary as products ($C_{products}$), transfer to a following process as intermediates (C_{inter}), and C in wastes (C_{waste}) and emissions ($C_{emissions}$). Then, the *productive* carbon is calculated by adding up $C_{products}$, which equals the difference between C_{in} and the C in wastes, emissions and intermediate products (equation 1). The *productive* C related to the originally fixed C_{in} is expressed as a percentage share for each (sub-)process (equation 2). In the fifth step, the *Carbon Utilization Degree* of the chain is calculated by adding up the *productive* carbon shares of all sub-processes (equation 3.3). In the final step, sub-processes with wastes and emissions or the complete chain can be analyzed further to identify the optimization potential and —optionally— to check whether compliance values are reached. The latter could be set by policy in the future, for example, that a minimum *CUDe* of 66 % has to be reached for a specific technology to receive tax reductions or to apply for incentives.

CUDe can be calculated according to equations (3.1)–(3.3):

$$C_{productive\ i}\ [\text{kg C}] = C_{products\ i} = C_{inter\ i-1} - C_{emissions\ i} - C_{waste\ i} - C_{inter\ i}$$

with $i = 1 \ldots n$; $C_{inter\ 0} = C_{in}$; $C_{inter\ n} = 0$ (3.1)

$$C_{productive\ i}\ [\%] = \frac{C_{productive\ i}}{C_{in}} \cdot 100 \text{ with } i = 1 \ldots n \quad (3.2)$$

$$CUDe\ [\%] = \sum_{i=1}^{n} C_{productive\ i}\ [\%] \quad (3.3)$$

C_{in} is the carbon content [kg C; C_{in} >0] of the entire harvestable biomass (*i.e.*, including the harvest residues) that enters the biomass transformation chain at sub-process i=1. It explicitly includes the C in harvest residues that remains in the field, for instance, as stubble, to address other sustainability aspects. Although C in stubble is finally returned to the atmosphere via soil biota on varying time scales, we consider it *productive* because it contributes to sustainable agricultural management. However, this effect is limited and could be accounted for more precisely, for instance by inclusion of site-specific characteristics.

C_{in} can be calculated from own data generated by chemical analysis, from published C contents of biomass that are available in the literature (some dry matter contents are listed in Table 6.14), or from data repositories, for instance, from ecoinvent [27]. Data on harvest residues can also be derived from repositories, for example, from FAO [28].

The total number of transformation processes in the conversion chain is denoted by n, whereas i denotes the respective sub-process, where C is further transformed to products and intermediates or is lost. $C_{emissions\ i}$ consists of the gaseous C losses [kg CO_2, kg CH_4] that are converted into kg C according to their molar conversion factors: CF_{CO2} = 12/44 [kg C/kg CO_2], and CF_{CH4} = 12/16 [kg C/kg CH_4] as well as fluid C losses. $C_{waste\ i}$ is the C in production waste. $C_{productive\ i}$ is the *productive* C of sub-process i, whereas *CUDe* represents the *productive* carbon of the complete transformation chain. $C_{inter,i}$ [kg C] is the carbon that is transferred as an intermediate product from sub-process i to the following sub-process i+1. We assume that every transformation chain yields some useful carbon. Hence, *CUDe* ∈ (0,∞], representing that C reuse is *theoretically* infinite. The upper frontier ∞ originates from the possibility of using biomass in a cascading way: Biomass can—like many non-renewable resources—be used several times, first (or more often) as a material and finally as an energy carrier. With this understanding, we follow the definition of cascading use in Carus *et al.* [29]. n greater than two means that after the harvesting step, (part of) the biomass is used at least two times. In such cases, in contrast to common productivity or efficiency calculations, the numerator can take values higher than the denominator and the total *CUDe* can yield values greater than 100 %.

In the following section, we apply the *CUDe* concept to simplified systems of current technologies that transform biomass into energy (bioelectricity from maize silage) or to a material (hemp fibers as insulation).

6.3.3 Example Application

Carbon Utilization Degree of a Biomass Transformation to Bioenergy—Anaerobically Digested Maize

Electricity generation from digested maize silage is a bioenergy pathway that is frequently associated with GHG mitigation potentials (e.g., 15–44 % of emissions compared to fossil electricity) [30]. However, it is necessary to be aware that some carbon is not productive along the biomass transformation chain (Figure 6.9).

Maize plants fix atmospheric carbon in their biomass. Biomass in the roots, leaves, and the lower part of the stem (stubble) remains in the field after the 'harvest' step, and its C content is returned to the soil pool, maintaining soil productivity. Ratios of 1–3 % of C in stubble *vs.* C in directly harvestable biomass have been reported from a long-term field experiment of different fertilizer treatments in maize [31]. Although C in stubble is eventually returned to the atmosphere via soil biota activity on varying time scales, we consider it *productive* because it contributes to sustainable agricultural management. Accordingly, it is included in the *CUDe* calculation. A total of 98 % of the harvested biomass is then transferred to the next step.

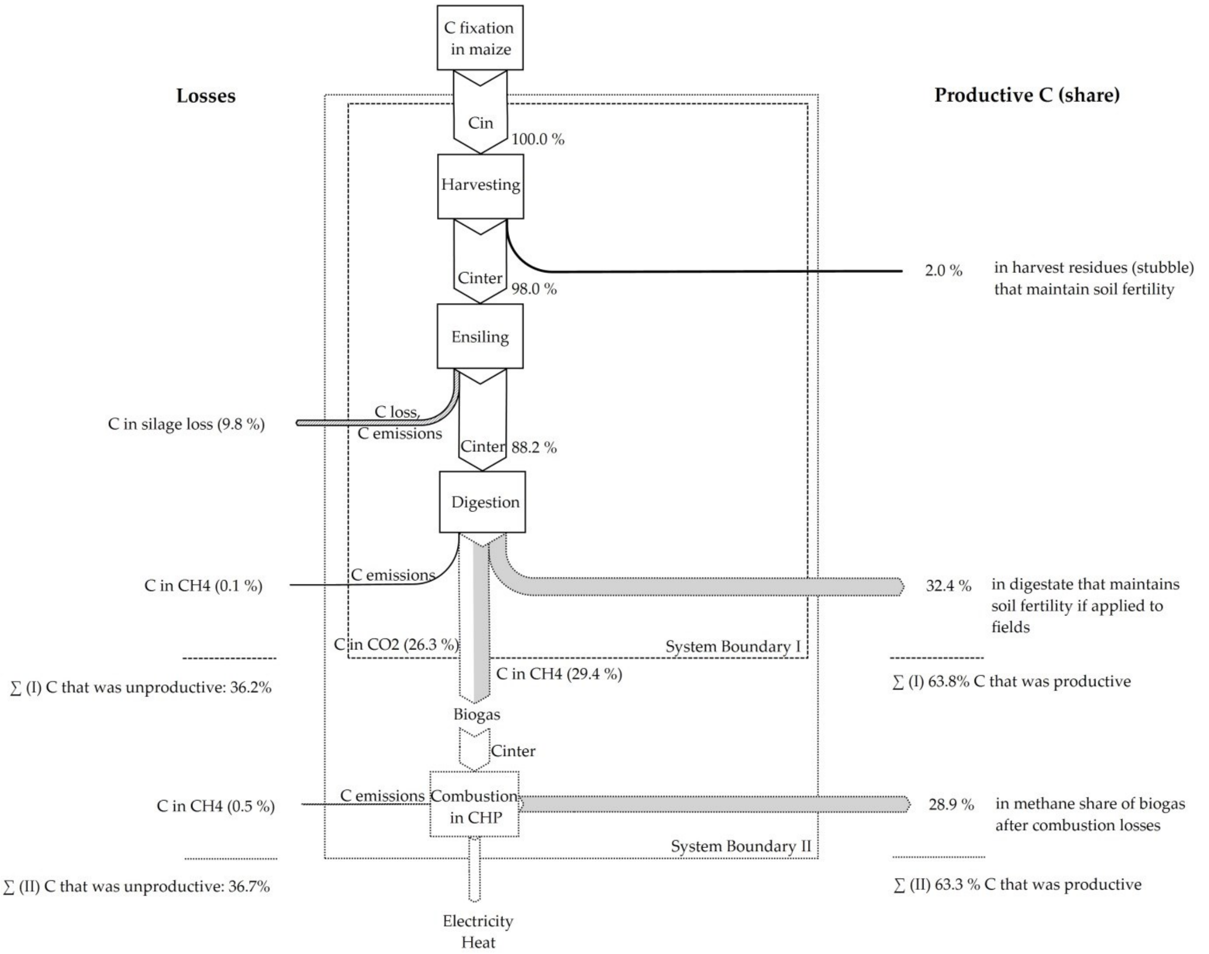

Figure 6.9: Carbon flows as a percentage of carbon fixed in harvestable biomass C_{in}, including stubble, resulting *productive* (grey arrows) and unproductive C (hatched arrows) during biogas generation from maize (Boundary I) and further use of this biogas in a CHP unit (Boundary II)

During the next step, ‘ensiling’, C can be lost in silage effluent as well as from microbial activity in the silage. Such losses have been reported with ranges from 1 to 3 % [32] as well as from 15 to 25 % [33]. We applied a value of 10 % loss. The maize silage is then transferred as an intermediate product to the ‘digestion’ process, where some gaseous leakage may occur (C lost as methane; 0.01% v/v [34]). The digestion step delivers digestate as a co-product, which can be re-applied to agricultural fields as a fertilizer, returning to the soil C pool and maintaining soil productivity [35], and can thus can be considered *productive*. Adding up the *productive* C for these steps of the biogas generation technology results in a *CUDe* of 63.8 % (*CUDe*=2 %+32.4 %+29.4 %; Boundary I in Figure 6.9) if we assume that only the CH_4 share of the biogas is of interest and optionally *productive*. If the boundary is expanded by including ‘energy generation’, the biogas is considered as an intermediate product. In a combined heat and power plant (CHP), the CH_4 share of the biogas (approximately 53 % v/v [36]) is burned to generate electricity and heat and therefore becomes *productive*. However, depending on the CHP engine type, 1.5–3 % of the methane may be emitted to the atmosphere [37]. The CO_2 share of the biogas (47 % v/v; [36]) is not considered *productive*. Accordingly, the total *CUDe* of the electricity and heat generation from maize silage results in a *CUDe* of 63.3 % (*CUDe*=2 %+32.4 %+28.9 %; Boundary II in Figure 6.9). More than one-third of the harvestable C did not become productive.

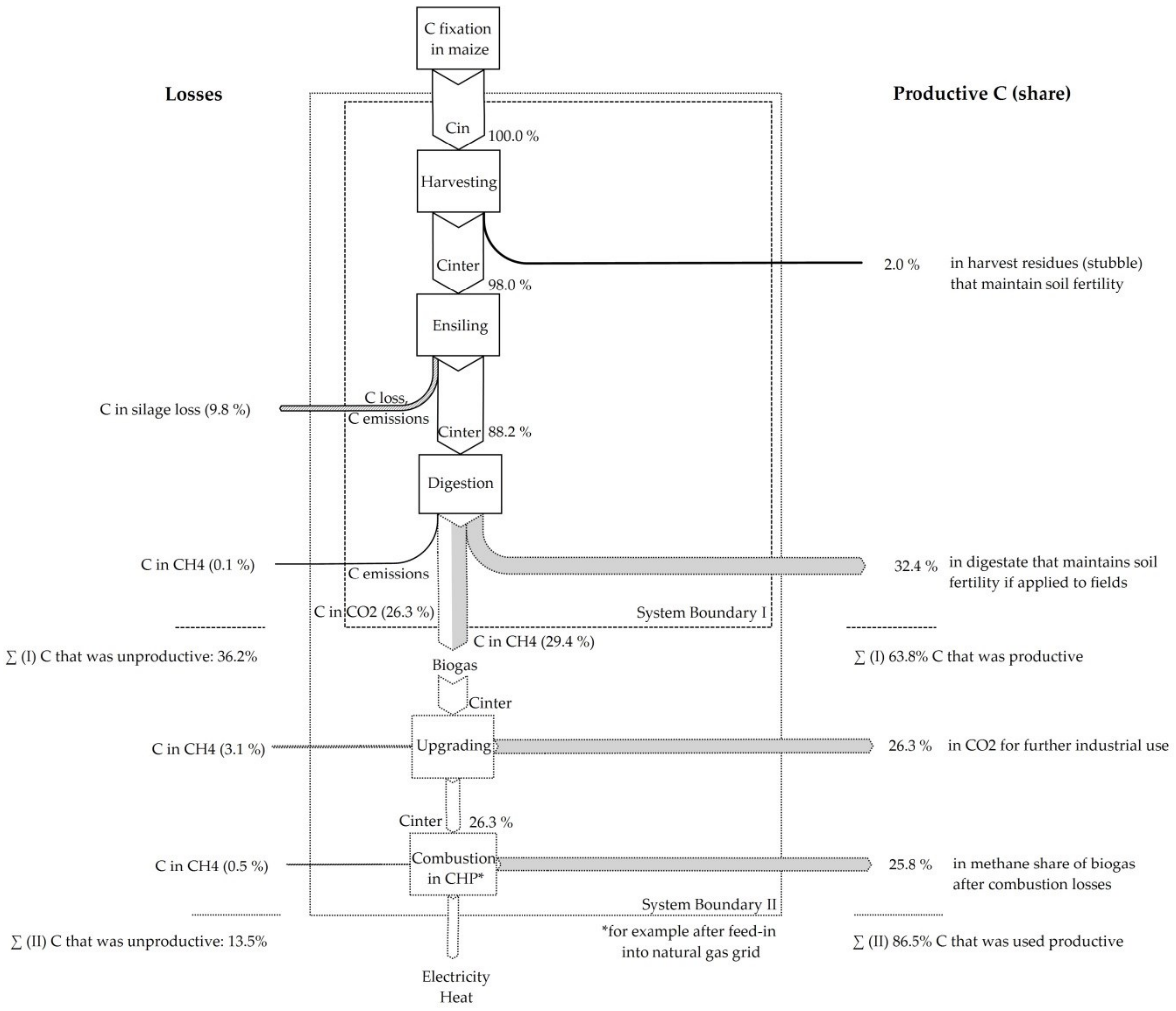

Figure 6.10: Carbon flows as percentage of carbon fixed in harvestable biomass C_{in}, including stubble, resulting *productive* (grey arrows) and unproductive *C* (hatched arrows) during biogas generation from maize (Boundary I) and further upgrading to bio-methane by conversion in a CHP unit, as well as separation of CO_2 for further industrial use (Boundary II)

Approximately 25 % of harvestable C_{in} is used by the microorganisms in the digester to metabolize the biomass to biogas, which as a consequence consists of a mixture of combustible methane and CO_2. If this CO_2 is separated from the biogas in an additional step to produce bio-methane (upgrading), the *CUDe* does not automatically increase. It could even decrease because of additional losses from 0.1–8 % during the upgrading, depending on the treatment process [38]. However, it could increase if the cleaned CO_2 share is used as a resource in further technological processes [8,39,40]. As the following example shows, upgrading biogas to bio-methane and utilization of the separated CO_2 could increase the overall *CUDe* up to 86.5 % (Figure 6.10). We assumed a feed-in into the natural gas grid and final use in a CHP plant.

Carbon Utilization Degree of a Biomass Material Usage—Hemp Fibers as Insulation Material

The overall *Carbon Utilization Degree* may increase for some technology chains with a cascading type of biomass use (Figure 6.11): Hemp is used as a material for building insulation (boundary I; *CUDe* =20 %+2 %+10 %+63 %=95 %), and after detaching, it is once again used as an insulation material.

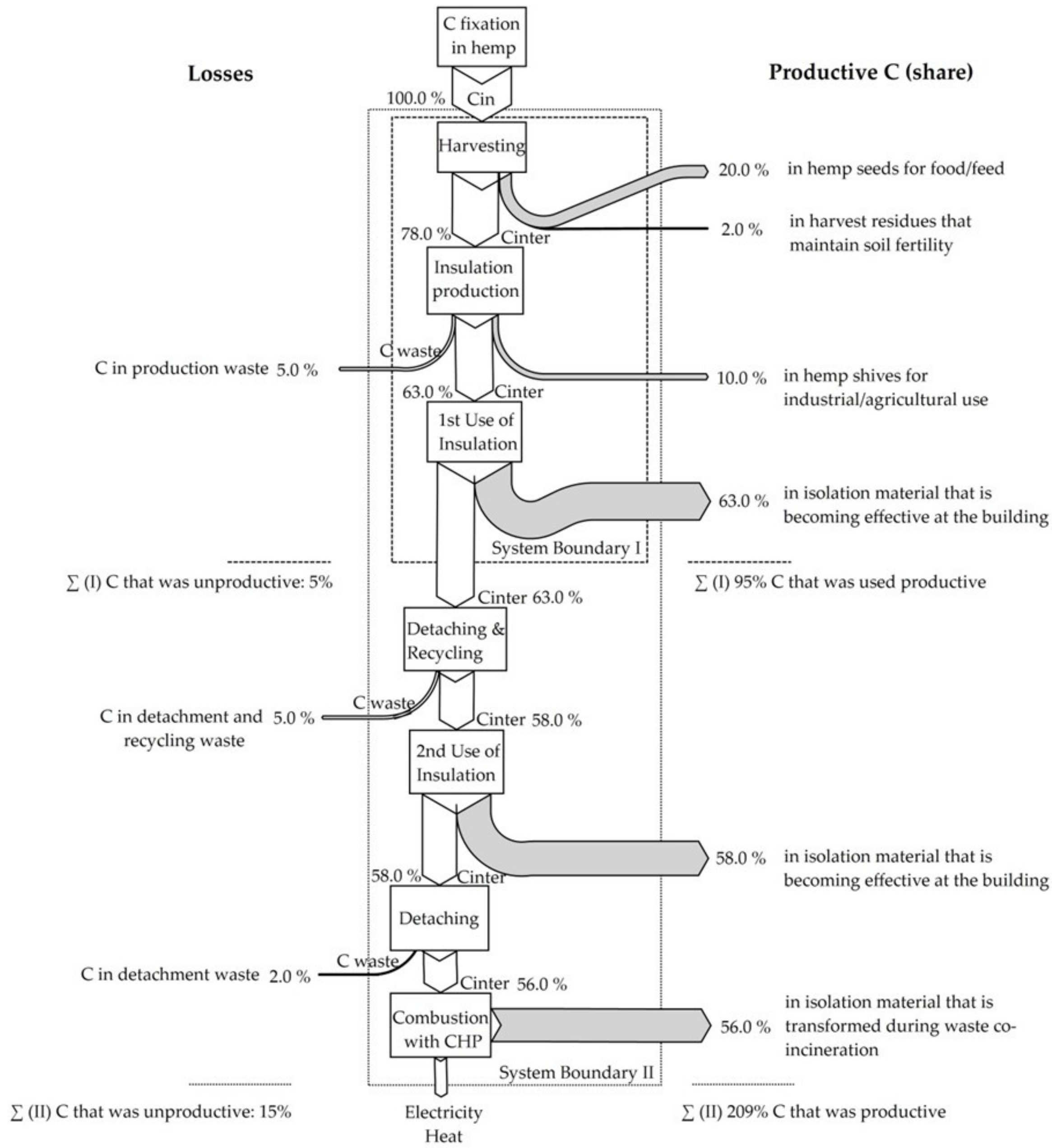

Figure 6.11: Carbon flows as percentage of carbon fixed in harvestable biomass C_{in} and resulting *productive* (grey arrows) and unproductive C (hatched arrows) of a cascading use of natural fibers as building insulation, followed by thermal recycling in a CHP unit

Finally, the material is detached and incinerated with energy recovery (boundary II). In this scenario, *CUDe* might reach 209 % (*CUDe*=20 %+2 %+10 %+63 %+58 %+56 %) for the transformation chain. Such a *CUDe* value greater than 100 % represents cascading C use which is explained at the end of section 6.3.2.

6.3.4 Discussion of the Approach

Impact-oriented approaches vs. resource-use-oriented approaches for policy decision support

Numerous assessment approaches have been published within the last twenty-five years that address carbon and that inherently use a productivity concept. Some of their specifications are listed in Table A1[12] and are compared to our *Carbon Utilization Degree* approach. Mainly, they address the sustainability goal of 'avoiding climate change' and thus are by definition impact-oriented assessments: Emissions of CO_2 and other GHGs are to be reduced because of their negative effects if they are released into the atmosphere. However, their names and/or methodological approaches suggest that they are productivity-oriented.

To achieve the sustainability goal of 'avoiding climate change', policy makers can choose between different regulatory methods: emission pricing (carbon taxes or 'cap and trade') or technology mandates and performance standards [41]. Some policy instruments are currently in place for carbon pricing, for example, the European Union Emission Trading Scheme (ETS) [42]. So far, mainly energy-intensive sectors, such as power generation and manufacturing industries, participate in the scheme. During the transition to a bio-economy, bioenergy may be included in the carbon trading market, calling for a reliable assessment of its CO_2 emissions. A pricing approach transforms C as CO_2 into a limited resource. For both pricing and standards, we need profound knowledge of the biogenic carbon emissions associated with the biomass conversion processes. It would not be appropriate to assume carbon neutrality of biomass (setting its emission factor to zero) or to calculate CO_2 emissions from the biomass C content. This has been discussed comprehensively in the scientific literature on biofuels [15,43,44]. Additionally, emissions from biomass conversion can vary depending on the type of biomass used, the process conditions and the conversion technology or emission reduction measures implemented [45]. Admittedly, this is also true for fossil-based energy carriers.

If we consider the difficulties to reliably assess biogenic process emissions and that additional criteria need to be taken into account to ensure that the transition to bio-economies is performed in a sustainable way (*i.e.*, not only addressing climate impact), then we should focus on the other policy options, technology mandates and performance standards. This is even truer because recent projections suggest that European targets—set at 40 % emission reductions compared to 1990 [46]—will probably not be met by the current policies (e.g., by the ETS, which is a pricing instrument [47]). If in society in general a transition could be initialized to improve efficient C use, *i.e.*, paradigm shift to 'C is a resource' from 'C is a threat', then more actors could enter the field to achieve the goal [48]. Such a paradigm shift by implementing efficiency standards for (biomass) conversion technologies could be a promising way to develop a sustainable transition pathway. Additionally, the strategy could go hand-in-hand with other public goals to increase efficient resource use [49] and energy efficiency [50].

Reliable criteria and appropriate indicators are necessary for such standards. To fill this gap, we proposed the *CUDe* approach. Optimization options could be identified at the process level, which subsequently could have an impact on the design of entire transformation chains. For bioenergy, the *CUDe* could offer a regulatory instrument, for instance, if a *CUDe* level exceeds a specific threshold, then incentives are paid, or fees fall due if a level is not reached.

Even if *CUDe* as an indicator might not influence policy making directly, it could have the potential to open debates and perspectives, which recently was identified as one important characteristic of indicators [51]. On the other hand, Runhaar [52] recently stated that the performance of integration tools is modest ("tools that aim to steer particular actors in such a way that they are stimulated (or forced) to incorporate environmental objectives in their policies or practices") and expectations should be realistic. Nevertheless, we think that *CUDe* could complement the existing assessment approaches toolbox as an additional indicator in a way that a 'dashboard' is provided, where different indicators are presented (as suggested by Jakob & Edenhofer [53]). Furthermore, a combination of integrated assessment models with those of other disciplines was recently identified as necessary to support policy formation and action toward low-carbon transitions [54]. As with the concept of 'umbrella' species that was proposed in conservation biology in the 1980s [55], *CUDe* could help to address more than one sustainability goal—avoiding climate change—because it inherently considers the enhancement or at least maintenance of soil productivity.

[12] Available as Table 4.6 in this document

Boundaries, Time Frames, and Carbon Sequestration

An important aspect of the *CUDe* approach is the definition that the C baseline is set at the carbon content of the theoretically harvestable biomass in the field. This addresses the aspect that input levels in agriculture are site-dependent (climate, soils, etc.). It is not our focus to advise where (and how) to produce biomass(-C) but to advise how we should use it. Methodologies are already available that are more suitable to choose biomass production ways, for example Life Cycle Assessment (LCA, [56,57]).

The *CUDe* system boundary includes all possible co-products in the analysis that a crop might yield. It also accounts for the fact that in the future, new technology options or market situations might be available to make the C in the harvest residues economically useful. Furthermore, this boundary enables, to some extent, the inclusion of ecological effects in the assessment, for example, the impacts on the humus balance and soil productivity. A prominent example is the use of straw, which could either be left in the field to, among other effects, replenish soil organic carbon pools or be used in stables for bedding or as an energy carrier for combustion [58]. In either use, the C content of the straw would be considered *productive*.

Another example of the ecological effects is the ecosystem service 'provision of important habitats'. In forestry, stubble use has been propagated in GHG mitigation studies [59]. This could trigger a loss of important habitats. Our baseline choice might reduce this pressure because the C in stubble is already considered *productive* and *CUDe* would not increase further.

The end-point of a *CUDe* analysis is not fixed, and it can be extended depending on the cycles of biomass use if the technology under study starts to use the carbon from biomass in a cascading way (as in section 6.3.3 *Carbon utilization…*). *CUDe* values greater than 100 % indicate cascading usage. The same effect has been reported from a cascade factor in the wood industry [60]. One could argue that additional energy—which is mostly C-based today—is necessary for C recycling. As already highlighted, biomass transformation systems should be assessed with a variety of metrics including energy-related ones, such as cumulated energy demand [61]. Hence, the concept could in the future be expanded by a combined presentation with such an energy-related metric, for example in a 2-dimensional metric to illustrate different biomass technology pathways and visualize target corridors.

Another relevant boundary is the time frame. Fixed time frames are defined in other approaches, for example, in the *Carbon Stability Factor* (CSF) for biochar [62] (100 years, Table A1[13]). For GHG assessments, different time horizons are used depending on the scope of the study and the longevity of the involved greenhouse gases. The published global warming potentials (GWP) with horizons of 20, 100 or 500 years reflect this [63]. These GWP characterization factors have been changing over time due to progress in the scientific understanding of atmospheric processes. The *CUDe* approach does not have a fixed time horizon by definition and, accordingly, does not rely on such external factors and is robust against changes in external metrics. Calculations of *CUDe* can be performed for different time horizons, but they must be properly communicated.

Biomass carbon can be stored in different pools with different time frames. In the context of climate change mitigation, the sequestration effect is an important aspect. However, the *CUDe* approach does not explicitly focus on this topic. This can be observed by how the C in soils is addressed. *CUDe* considers the C, which is returned to the soil, as *productive* (e.g., it could improve soil fertility), even though it is eventually re-emitted to the atmosphere by soil biota activity. This represents the perception that *CUDe* is an approach for efficient C use in general, not just with a focus on climate change mitigation. In the latter case, it would be necessary to account for additional benefits for C that is stored long-term.

Multi-product systems, such as most biomass conversion systems, can be assessed by numerous approaches. The methodologies account for possible product and co-product diversity. For instance, LCA, as an impact-oriented assessment, uses, among others, 'system enlargement'. However, system enlargement can lead to increasing uncertainty in the analysis' outcome due to the diversity of possible biomass uses and potential reference products. *CUDe* considers all biomass co-products in its

[13] Available as Table 4.6 in this document

calculation directly; hence, it avoids the difficulty of defining reference products and reduces the time for the analyses because no additional data need to be gathered.

The *CUDe* approach could help to compare biomass transformation systems where biomass is used for energetic and/or material purposes. Although different biomasses have similar C contents per dry matter (Table 6.14), they can lead to differing *CUDe* values as one biomass can be more suitable for a certain purpose than another. Thus, the approach considers different biomasses as well as the design of biomass conversion chains as a whole.

6.3.5 Conclusions and Outlook

Existing approaches to assessing C, which are used to analyze biomass conversion chains, have some critical issues to address. These include external effects, such as changes in the underlying assumptions. Robust indicators for decision support for biomass use are needed. We proposed *Carbon Utilization Degree CUDe* as an indicator that represents the efficient use of carbon as a production factor in biomass conversion processes for energetic and material use. This indicator could reflect a paradigm shift that CO_2 is not a threat but a finite resource that requires suitable management. *CUDe*, as a supplementary indicator for existing methods, could aid in the design of policies for biomass transformation pathways by defining threshold values for efficient carbon use in conversion processes. The approach needs additional testing to prove its applicability even to more complex pathways than those provided in this manuscript.

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Author Contributions

Anja Hansen and Jörn Budde developed the idea and the methodological concept; Anja Hansen prepared the case studies, figures and wrote most of the paper. Jörn Budde, Yusuf Nadi Karatay and Annette Prochnow contributed to the discussion of the concept and to the organization and phrasing of the manuscript.

Conflicts of Interest

The authors declare no conflicts of interest.

Abbreviations

The following abbreviations are used in this manuscript:

C: Carbon
CF: Characterization factor
CH_4: Methane
CHP: Combined Heat and Power
$C_{emissions}$: Carbon in gaseous losses, e.g., in CO_2 or CH_4
C_{in}: Carbon content in biomass dry matter that enters the biomass conversion chain
C_{inter}: Carbon in intermediate products
CO_2: Carbon dioxide
$C_{productive}$: Carbon that becomes productive in a wide anthropocentric view
$C_{products}$: Carbon in final products and co-products
CSF: Carbon Stability Factor
CUDe: Carbon Utilization Degree [%]
C_{waste}: Carbon in waste, e.g., in production waste
EU ETS: European Union Emissions Trading System
GDP: Gross Domestic Product
GHG: Greenhouse Gas(es)
GWP: Global Warming Potential(s)
L.: Carl von Linné (botanical author citation)
LCA: Life Cycle Assessment
MACC: Marginal Abatement Cost Curves

NPP: Net Primary Production
NEP: Net Ecosystem Production

Appendix A

Table A1. Overview of some productivity approaches dealing with carbon. CUDe—Carbon Utilization Degree, CSF—Carbon Stability Factor, GDP—Gross Domestic Product, GHG—Greenhouse Gases, MACC—Marginal Abatement Cost Curves, NPP/NEP—Net Primary Productivity/Net Ecosystem Productivity, S&P/IFCI—Standard & Poor's International Finance Corporation Indexes.

Online http://www.mdpi.com/2071-1050/8/10/1028/htm (and as Table 4.6 in this document)

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7 Discussion

7.1 Need for a Systemic Approach of Biomass Usage for Climate Change Mitigation

Climate Change mitigation is a challenge that should be met by an "all hands on deck" approach (Anonymous 2016). It is therefore necessary to involve as many actors as possible. However, each actor can only perform where he/she has the opportunity to do so. This is sometimes neglected by society which for example has called for sustainable agricultural production but associated with it some aspects which are out of the scope of farmers like land use planning, storage losses within supply chains, impacts from processing, retail or households, reductions in animal product consumption or omission of biofuel quota (Hansen & Wolf 2015). Integrating the actions along the different actor levels from producer over industries to consumers is therefore important and must be accomplished with reliable tools to first identify and finally monitor the most promising actions.

Numerous tools are available, many of them following an understanding of life cycle thinking that has been put forward by the LCA methodology. However, allocating impacts to different processes in agricultural production chains has been found challenging, because in agriculture, natural cycles and industrial processes are intertwined and therefore difficult to assign to natural or human activities separately (Möhlmann *et al.* 2000).

The assessment of climate change mitigation with bioenergy evolved over decades starting in the 1990s (see for example Garrett 1992; Hall *et al.* 1992; Ellington *et al.* 1993), and still consensus is missing on some points as for example appropriate baselines (Sanderman & Baldock 2010; Johnson & Tschudi 2012; Soimakallio *et al.* 2015; Brander 2016) or time frames (Cherubini *et al.* 2011; Yuan *et al.* 2015), the latter not being addressed in detail in this dissertation. The success of mitigation measures needs to be analyzed in the acceptance of fuzziness. It should not only consider a narrow agricultural context but instead follow a broader understanding of agriculture as a sector that consists not only of activities on fields/in stables but involves processes down to the consumer and his understanding of sustainability and of an adapted life style. As well, decisions need to be made in acceptance of and despite of fuzziness, for example that it cannot be taken for granted that bioenergy reduces climate impacts.

7.2 Uncertainty and Communication

Fuzziness needs to be communicated in a way that does not give the impression that everything is unclear which leads –with high certainty– to inaction. Despite massive scientific evidence on human-induced climate change (IPCC 2014), some skepticism is still existent in public discourse (Poortinga *et al.* 2011; Kaiser & Rhomberg 2016), or among farmers (Asplund 2016). The authors of the Intergovernmental Panel for Climate Change (IPCC) agreed on a way how to communicate the degree of certainty of the findings that are presented in IPPC reports which they called 'calibrated language'. In a guidance note for lead authors of AR5 (Mastrandrea 2010), two metrics to communicate that degree were proposed: qualitatively expressed confidence in the validity of findings and probabilistically expressed quantified measures of uncertainty. Whereas these guidelines have been followed more or less strictly by the AR5 authors, a follow-up study on English-speaking news media revealed that the approach had been adopted only by 14 % of in total 1900 journalists' reports on the AR5 findings (Collins & Nerlich 2015). Instead, analogies to other scientific principles or examples of 'taking action despite uncertainty' have been used by journalists, politicians and scientists to communicate results. The study concluded that science needs communication strategies that account for the different levels of understanding in their respective audience. In that regard, the scientifically/mathematically correct representation of our knowledge of

- how reliable mitigation potentials are or
- how probable it is to mitigate CC with the implementation of bioenergy pathways and an increased use of biomaterials,

is not necessarily enough to engage society and politicians to install and pursue mitigation measures. Also a thoughtful visualization with pictures of CC causes, impacts, or solutions is required. Otherwise, the intended effect of motivating actions might not be obtained but instead cynicism might be provoked (Chapman *et al.* 2016). Considering these difficulties, it might be recommended to address CC not as a singular issue but combined with other challenges. Achieving results for several aims jointly, is a familiar *win-win* concept to many people. In that regard, it might help to switch from looking at carbon containing gases (CO_2 and CH_4) only under the topic of CC, but instead as an optimization task for matter flows and nutrient cycles. The same is true for the effective greenhouse gas N_2O which is as well embedded in essential nutrient cycles and, furthermore, is intertwined with the carbon cycle (Gruber & Galloway 2008; Robertson *et al.* 2011; Lehman & Osborne 2016).

One could argue that the purpose of science is not to inform policy with easily digestible pieces and provide ready-to-go solutions. How science can inform and support policy is a field in its own, (see for example Braat & de Groot 2012; Scheer 2015), that needs to consider that a complex topic as climate change must be addressed and communicated jointly by different disciplines (Hallegatte & Mach 2016).

The need to address uncertainty, for instance with MC or scenario analyses, is increasingly acknowledged in science, and seems to become common practice in mitigation studies (Table 4.4, Table 7.1). In public perception, this is not yet generally admitted (see section above), or understood.

In public, not absolute uncertainty of mitigation potentials seems to be of main interest but especially those single uncertainties which have the highest impact on study results. The LUC issue is an example which swept from the biofuels debate (Searchinger *et al.* 2008) to other agricultural commodities, as for example livestock products (de Vries & de Boer 2010). In that regard, more case studies are needed to identify other possible uncertainty hotspots, and, on the other hand, to evaluate if the already known uncertainties are relevant for other regions or production systems as well. Harsono *et al.* (2012) for instance showed the GHG effects of different scales of palm oil production for biodiesel. In existing studies on bioelectricity from woody biomass, relevant aspects of uncertainty in the calculation of GHG emissions and related emission savings (Table 7.1) have been analyzed, whereas seldom jointly in one study: SOC, technology choice (for instance gasification efficiency), reference technology and feedstock, baseline, yield. Modelling uncertainties were sometimes found less important than parameter uncertainties (Malça & Freire 2010), contrary to results from case studies one and two (Hansen *et al.* 2013; Hansen *et al.* 2016a). As well, meta-analyses tried to identify possible ranges (Djomo *et al.* 2011; Kadiyala *et al.* 2016). With few exceptions, woody biomass was reported as an option to emit less GHG per electricity unit than from a fossil reference. Conversion technologies and their associated pre-processing had an important impact on results. Co-firing instead of biomass-only conversion for example increased GHG emissions nearly fivefold from 57.9 to 278.2 g MJ_{el}^{-1} (Kadiyala *et al.* 2016). Drying of pellets with fossil fuels doubled the emissions, and accounting for CH_4 emitted during storage increased the uncertainty (Röder *et al.* 2015). However, such emissions are quite uncertain, for example due to measurement challenges as described for wood chip storage losses (Lenz 2017). Integrating SRC cultivation into regional activities by combining wastewater treatment and energy generation yielded comparable mitigation potentials (>50 %) to stand-alone generation chains, even though its GHG emissions were at the upper range (103 g CO_{2e} MJ_{el}^{-1}) (Buonocore *et al.* 2012).

In addition to standardized uncertainty presentation (Mastrandrea 2010), good-practice in study completion might as well increase credibility of results. Converting energy units from different studies (MJ to kWh and vice-versa), might lead to apparent ranges. This can be due to that no specific boundaries are given: MJ could be interpreted as MJ_{in} (energy of input biomass), as MJ_{out} (energy in final energy carrier), or as MJ in final energy (heat or electricity). This thesis related all energy units to final energy (MJ_{el} and MJ_{heat}). A similar problem might arise from unclear GHG specification: if just CO_2 is accounted for, or also other GHG (CH_4 and N_2O) which have been converted to CO_2 equivalents. This has emerged as a problem in the collection of carbon productivity approaches (Table 4.6). However, meanwhile, most studies comprise all three gases into their assessment. CH_4 uptake from soils is as well more often included (Robertson *et al.* 2011; Nikièma *et al.* 2012; Kim *et al.* 2016).

Table 7.1: Emission (E_B) and mitigation (MF_B) factors (g CO_{2e} MJ^{-1}, mitigation potentials MP_B (%) and reported uncertainties for bioelectricity from woody biomass (update to Table 6.7 (Hansen *et al.* 2013); sorted in ascending order of E_B where available)

Biomass Species / Conversion Technology	Emission Factor E_B [g CO_{2e} MJ^{-1}]	Mitigation Factor MF_B [g CO_{2e} MJ^{-1}]	Mitigation Potential MP_B [%]	Ref.
Wood chips in biomethane-gas-steam power plant	*n.a.*	-30-65	28 - >100	WBGU (2009)
Willow chips gasification (optimistic∣pessimistic scenario regarding machinery efficiency, N_2O emissions, C fixation in soil and biomass)	*n.a.*	62 ∣ 89	*n.a.*	Lettens *et al.* (2003)
Bioelectricity from poplar wood chip gasification	**12 (case study)** **-34±21(MC analysis)** **10±4 (neither C sequestration nor N_2O reference included)**	**294 (case study)** **274±21 (MC analysis)** **230±5 (no seq./ no N_2O)**	**n.a.**	**Hansen *et al.* (2013)**
Gasified SRC willow in CHP/Denmark [a,b]	0.8 (on previous cropland) -10.4 (on marginal pasture) -31.8 (on marginal abandoned land)		*n.a.*	Saez de Bikuña *et al.* (2016)
Wood pellets for Swedish heat and electricity from domestic and imported feedstock [c]	1.7-25.4	*n.a.*	64-98	Hansson *et al.* (2015)
Review of 26 GHG balance studies of willow and poplar based bioenergy	10.8-36.7	*n.a.*	*n.a.*	Djomo *et al.* (2011)
European mixture of local and imported biomass in different restrained pathways of electricity	38.6-65.4	*n.a.*	63.2-78.3	EEA (2013)
Wood pellets from forest ∣ saw mill residues	36.7 ∣ 38.9 (base case) 75.3 ∣ 77.5 (A: fossil fuels for drying) 88.0 ∣ 137.5 (B: CH_4 from storage) 225.5 ∣228.3 (A+B)	*n.a.*	83 ∣ 82 (base case) 64.8 ∣ 63.8 (A) 59 ∣ 34 (B) -8 ∣ -9 (A+B)	Röder *et al.* (2015)
Dedicated energy crops in biomass-only electricity generation systems (review of 19 studies incl. oil, starch and other feedstock)	57.8±87.9	*n.a.*	*n.a.*	Kadiyala *et al.* (2016)
Willow farming for bioenergy in an integrated wastewater treatment system in Sweden	103	108 [a] 205 (vs. coal-fueled CHP)	51 [a] 66 (vs. coal)	Buonocore *et al.* (2012)

[a] referenced to *EF* from natural gas CHP unit; [b] MC analyses results presented as error bars; [c] explicitly not from SRC but from saw dust and round wood; [d] 29 CO_{2e} ha^{-1} sequestered over 23 year plantation life time; [e] 20 % dry matter loss included; [f] 3 % of carbon in biomass as methane in storage losses; *n.a.* not available; CHP – Combined Heat and Power Plant

Leakage is another effect that contributed to the impression of uncertain mitigation by biomass usage in the public. It may arise if policies (or analyses) do not cover the complete scale of addressed effects and can take shape as a transfer of GHG emissions to other geographical locations, sectors, products or life cycle stages (Plevin 2010). It can also arise as market mediated effects: rebound effects may result from efficiency increases, which decrease prices and in turn increase consumption, or indirect effects such as indirect land use change. From the existing uncertainties, Plevin (2010) concluded that crop-based transportation fuels policies based on global warming intensity thresholds are no reliable way to mitigate climate change. Similarly, Franks and Hadingham (2012) argued that mitigation options at the farm level will not deliver targeted reduction levels from the agricultural sector and hence, policy should focus more on demand-side measures like carbon taxes. To avoid leakage, consumption-based accounting has been proposed to allocate emissions not to their place of occurrence but instead to their place of initiation (Feng *et al.* 2013). However, leakage could be considered less a methodological problem in itself but a problem of proper method application.

The most often addressed uncertainty in the context of biomass usage regards LUC effects. These consist of two correlated uncertainties: the absolute height of carbon stock changes which in turn can only be assessed in relation to a properly set baseline, which will be discussed in the following.

7.3 Baselines

Land use change –with the main effect of above-ground and soil organic carbon stock changes (4.1.5)– has been the most discussed uncertainty in scientific mitigation studies. Supplement 12.1 provides detailed points of this discussion with a focus on C sequestration under energy crop plantations. Assessing change calls implicitly for a baseline definition against which the change is expressed. Often, a positive rating of bioenergy and biomaterials is due to positive C stock effects which denote the possibility to reduce emissions and to increase sequestration as well. Also in this thesis, LUC effects were taken into account and were relevant for the assessment of bioelectricity (dLUC, Hansen *et al.* 2013) as well as for the ranking of systems (dLUC, iLUC Hansen *et al.* 2016a). Both analyses dealt with SOC changes on a more general level, whereas increasingly experimental studies identify distinct biomass C sources on the field scale and how they allocate into different stocks in more detail. Carvalho *et al.* (2016) for example found that in *Miscanthus* the aboveground biomass C allocation to deeper soil horizons is higher than for other bioenergy crops as maize or sugarcane (*Saccharum officinarum*). Hence, such plant characteristics could be a criterion to choose between different crops. For species grown under SRC management such as poplar, plant traits that influence such allocation have as well been stated (productivity, biomass distribution to roots, and fine root/coarse root ratios) (Garten Jr *et al.* 2011). The allocation characteristic is important in the context of the duration of sequestration effects: the deeper the C is allocated into the soil the less the probability that it will be re-activated after perennial crops are grubbed and cultivated with annual crops. However, substantial scientific evidence is still missing for such effects because often C contents are only measured in the upper soil layer (Schmidt *et al.* 2011). As well, not many SRC plantations have yet reached their depreciation age and accordingly, long-term measurements are missing.

Besides the uncertainty effect on local assessments, at a global scale, the aboveground C pools in vegetation in different biomes may become more important with regard to climate variability and impact on global C cycles vice-versa, as was shown by the analyses of La Niña effects in Australia (Poulter *et al.* 2014). They found that semi-arid biomes were acting more as drivers of the C cycle than the previously most relevant tropical forests. Overestimated C turnover rates between plant-soil pools in global models were found to result in an overestimation of soil C sequestration potentials and consequently in misjudgment of soils as C sink (He *et al.* 2016). In another modelling approach, initial C content of sites used for plantations were eventually considered as relevant for the C sequestration potential (Garten Jr *et al.* 2011).

Studies on biomass potentials (and on fossil fuels and critical metals) are often associated with relevant uncertainty (Speirs *et al.* 2015). Still, potential studies for bioeconomy often consider 'agricultural residues' as an important feedstock pool (Batidzirai *et al.* 2012). Their intense removal, for example for use in cellulosic biofuel chains, might deplete SOC stocks on the long run as soils are cut off from subsequent supply of organic C (Carvalho *et al.* 2016). With this in mind, the *CUDe* approach (Hansen *et al.* 2016b) considered C in plant residues already as productive (6.3.2) in order to avoid increased pressure on that C pool.

Increasingly, also other gases than CO_2 from plant and soil pools are considered in the discussion of baselines, N_2O being the most frequently addressed which is further discussed in the paragraph below. Others are CH_4 or volatile organic compounds (VOC). The latter can have an impact on Earth's radiative balance through generation of aerosols, and have negative health and yield effects as well. For isoprene, Morrison *et al.* (2015) found higher fluxes from SRC willow than from annual crops (wheat, rape). Ashworth *et al.* (2013) modelled isoprene emissions, their effects on wheat yield reductions and on mortality, from a SRC implementation that would meet European biofuel targets. Even though small, such emissions would importantly counteract ozone-related pollution control policies.

For biomass GHG assessments, the experimental reference site could be established in a paired-plot design at the adjacent arable field to safeguard similar climate and soil conditions (Laganiere *et al.* 2010). However, such designs are not yet very common in recent bioenergy evaluations that include N_2O emission studies, for example are not applied in Zenone *et al.* (2016). Kim *et al.* (2016) reviewed emissions from agroforestry systems and adjacent fields, of which one study reported a N_2O emission rate of 1.4 kg N_2O ha^{-1} yr^{-1} from a tree plantations on arable land with reference emissions from an adjacent field twice that high. Oates *et al.* (2016) as well reported higher N_2O fluxes from annuals compared to perennials (4.9-30.0 kg N_2O $ha^{-1}yr^{-1}$ and 1.7-9.9 kg N_2O $ha^{-1}yr^{-1}$, respectively). As well did Drewer *et al.* (2012), depending on fertilization levels (wheat, rape vs. *Miscanthus*, willow). The absolute values are twice the rate considered in Hansen *et al.* (2013), the ratio between SRC and cropland emissions height being similar. Meurer *et al.* (2016) collected N_2O emission data for Brazilian land use types and found that previous land use had an important effect on emissions whereas soil properties had not. N_2O emissions from cropland used for unfertilized agroforestry were approx. 1.2 kg N_2O ha^{-1} yr^{-1} which is in the upper range of emissions of the unfertilized poplar plantations in Hansen *et al.* (2013). LU types were not evenly distributed between all biomes, so no values for all LU could be derived. Dechow and Freibauer (2011) found N-fertilization being the relevant driver for regional N_2O emissions for perennial land use 'grassland', whereas emissions from cropland were highly influenced by climatic conditions and soil properties. CH_4 emissions were reported as negligible (Drewer *et al.* 2012), or not significant (Nikièma *et al.* 2012). The latter authors found less CO_2 but higher N_2O emissions from SRC than from the reference pasture.

Which one the appropriate land use base line is in impact assessments of land-derived production systems, is being discussed (Soimakallio *et al.* 2015; Brander 2016) (4.1.5). Soimakallio *et al.* (2015) consider 'natural regeneration' as the appropriate baseline option –even though they consider it unrealistic in most situations– for analyses where accounting starts from human-induced land use. Hansen *et al.* (2013) chose 'business as usual'(BAU) as the baseline. In this case study, BAU cultivation in the human-induced land use included no fertilization, i.e. no further human interventions, and hence, represent natural emissions that derive depending on local interactions between crop, soil and climate. Such N_2O credits altered the bio-electricity from poplar wood chips from a low-emitting energy source to a carbon neutral[14] or minor sequestering energy source if considering uncertainties (Table 6.5). Brander (2016) generally criticized the approach of using natural regeneration baselines and argued that it must be distinguished between 'foregone sequestration' (which is to be assessed via the net change compared to the baseline) and the allocation and timing of LUC emissions. In that sense, the approach followed in Hansen *et al.* (2013) (6.1) for the N_2O reference emissions from the annual crop rye fell into the group of LUC emissions which need to be allocated.

Related to the baseline discussion is the technology assumption of CCS implementation. Especially with the possibility of closing fast C cycles, bioenergy with CCS has been promoted as an option to substantially reduce atmospheric CO_2 levels (Venton 2016). Luckow *et al.* (2010) for example assumed that if CCS becomes economically feasible due to CO_2 pricing, biomass would preferably be transformed in stationary electricity generation, whereas without it would be used as transportation

[14] in the broader sense of CO_{2e}

fuel. Others see CCS more critically and categorized it as a hype (Martínez Arranz 2016). CCS was not considered in this dissertation, as it is not yet state-of-the-art in energy generation. As well, it would also be applicable for fossil energy generation pathways, and hence, would result in lower reference emission factors EF_F.

Any baseline application implies the possibility of negative net values (see also masking effect in Table 4.2). This is true on the plot/farm scale as well as on the system-wide scale. GHG mitigation analyses are per definition comparisons (usually) against a fossil reference, which is applied as a baseline at the end of the study. Especially in the agricultural context, additional baselines might necessarily be applied for unit processes of biomass production (see above discussion on N_2O). As agriculture occurs on land, and some emissions are calculated land-based, it is also required to concentrate one's attention on the appropriate identification and calculation of land demand to safeguard proper results for such land-dependent values.

7.4 Multi-criteria

GHG mitigation analyses are first of all –in the narrow sense– single-criterion assessments, as they compare the GHG emission of bioenergy versus the emission of an alternative (usually fossil, but other renewables or mixtures of different energy sources are possible). Meanwhile, alternatives to the fossil comparators have been proposed, for example to express the necessary wood/afforestation area to offset emissions from human activities as for example farm production (Torres *et al.* 2015).

Yet, climate change is not only single species-driven but a basket of different gases contribute to this effect. Hence, GHG mitigation analysis already is inherently a multi-criteria assessment if the different GHG gases are considered separately: along the bioenergy generation chain, its unit processes contribute to the different GHG emissions and offer different opportunities to optimize the process chain. However, optimizing a production chain for one gas might worsen it for another. In agriculture, carbon and nitrogen cycles (Gruber & Galloway 2008; Soussana & Lemaire 2014) are closely interlinked and accordingly are their GHGs (CO_2, CH_4, N_2O). One example is the stover removal from maize which could increase area-related energy yields but on the contrary might induce increased N_2O emissions (Lehman & Osborne 2016). Such drawbacks are usually avoided by balancing all emissions to one common denominator (CO_2 equivalents) (see 4.1). An example is the conversion of pasture to SRC plantations, where CO_2 emissions decreased, but as N_2O emissions increased, in total, a CO_{2e} debt was unveiled (Nikièma *et al.* 2012). Leakage effects (7.3) along the process chain can be avoided by proper boundary setting.

The call to consider more than a the single[15]-criterion 'CC mitigation' and to develop a holistic view has been stated repeatedly, for example from Wagner and Lewandowski (2016) for willow and *Miscanthus*) and others (Ulgiati *et al.* 2006; Foster *et al.* 2014). This concept is as well the basic idea of LCA. However, how many and which indicators are necessary for a sufficient assessment and at the same time for avoiding overparameterization is still being discussed. Steinmann *et al.* (2016) stated from an analysis of products from a common LCA data base (ecoinvent) that four to six indicators were enough as they covered 84-92% of variance in product rankings. Climate change and land use were two of them. In Hansen *et al.* (2016a), the three criteria LU, CC and fossil fuel demand were insufficient to clearly distinguish between the two strategies at hand, maybe due to CC and fossil fuel demand being associated.

Land use and CC impacts are usually considered unison, especially if LUC is reflected (De Rosa *et al.* 2016). The climate impact of LU/LUC can be addressed by existing impact factors, whereas for other environmental impacts as for example nutrient leaching, biodiversity changes and water resource depletion, impact factors are rarely available (De Rosa *et al.* 2016). Case studies (Immerzeel *et al.* 2014) and modelling exercises (Tarr *et al.* 2016) exist that address biodiversity in SRC and energy forests (3.1.1), however they have not yet been aggregated to generally applicable biodiversity impact factors.

[15] In the sense that CO_{2e} are considered, and not the different gases separately

Implementation of a productivity-based approach to assess carbon (Hansen *et al.* 2013) could indicate, where C usage efficiencies of existing technologies could be increased, accompanied by reductions in GHG. Similarly, this was shown from Losordo *et al.* (2016) for bioethanol efficiency by incorporating a biotechnology approach. For other second-generation biofuels such as bioethanol from Brazilian sugarcane, increasing N use efficiency (NUE) has also been identified as a way to improve biofuel production and reduce its environmental impacts (Otto *et al.* 2016).

In a cross-linked world, decision makers need tools to reflect possible impacts of decisions. It is risky to base decisions on just one single indicator. Numerous policy strategies have already been announced on the national as well as on the European level which aim at efficient resource use (European Commission 2011; BMEL 2013; BMUB 2016). They have been also brought forward in the awareness that with a switch to bioeconomy there will be a growing need to allocate biomass, land as well as fossil resources. Bioeconomy production chains usually consist of numerous interacting processes, which make system analyses an ambitious task. Life cycle assessment enables multi-criteria analyses even though, owing to the complexity of systems, the method needs very careful application. However, already the attempt to apply it helps to explore the cross-links in as well as in-between systems and helps to get indications on possible effects even though they cannot be illustrated with detailed, statistically confirmed numbers. Even if the call for multi-criteria assessments is loud, it is seldom consequently followed. In that sense, it might be constructive to inherently address multi-functionality and multi-criteria by a paradigm switch, and indicate it with a new language on working carbon (McDonough 2016).

8 Conclusions and Outlook

According to the results of the case studies, bioenergy, in particular bioelectricity from woody SRC biomass, could contribute with high agreement/medium evidence (terminology according Mastrandrea 2010) to climate change mitigation efforts.

Under specific conditions – (i) increasing soil organic carbon stocks (ii) or/and reduced N_2O emissions relative to a reference crop –, this pathway could as well result in fixation of some atmospheric carbon to longer-lasting C-pools. How much the soil carbon pool contributes to the mitigation remains uncertain due to missing long-term monitoring data. Net differences in crop species-related N_2O emissions would contribute to the mitigation effect with medium to high confidence which can be stated from measurements evidence. In consequence, biomass cultivation should be baselined against more than the agreed-on, but still uncertain, C stock changes: it should account also for crop-specific differences in N_2O emissions.

Ambitious mitigation calls for carbon-negative human activities wherever possible, indicated by negative emission factors, or mitigation potentials greater than 100 %. Biomass usage should be optimized in that regard. Such values depend as well on fossil reference technologies which are changing over time. Therefore, bioenergy choices have to be adapted to changing electricity mixes as well.

The reference system for crop cultivation for biomass usage can be based on a status-quo system if comprehensive data exist on (i) current equilibrium status of the land, (ii) current emissions. The proper baseline choice has as well to be made in a larger systemic context as well as on the specific local scale. It might be that ‘local’ uncertainties related to a single bioenergy generation may be less relevant at a broader scale. Additional case studies should evaluate such uncertainty hot spots.

Site dependency of agricultural production and downstream processes always call for evaluation of biomass usage for each specific application and regional context. Transfering results untested to other usage chains is not advisable as other impact hot spots exist in different regions or production systems. As well, other than already known uncertainty hot spots might be of relevance there. To account for that and to reach ambitious CC targets, emission thresholds set by policy could apply for electricity and heat in a general manner, regardless if bio-/solar or fossil energy-based.

The case studies in this dissertation focused on stationary bioelectricity generation. Such applications can exhaust the energy potential of biomass with sophisticated technological solutions. As well, integration of biomass usage into complex economic chains offer efficient use of biologically fixed carbon, and safeguard the reaching of ambitious mitigation targets.

How biomass should be allocated to its different possible uses was not the aim of this thesis. Starting point was if mitigation analyses are suitable to evaluate biomass usage and identify promising solutions for a low-carbon economy. In conclusion, it is necessary to consider biomass usage always as an integral part of a larger cross-linked system (the same is true for other resources), and to increase efficiency (besides the general statement that demand side reductions are necessary).

Biomass cultivation and subsequent transformation processes should be assessed by multi-criteria approaches, including mitigation analyses. At least three independent criteria are advisable.

Agriculture is the basic production sector for human society, and a sector that can act as a source as well as a restricted sink for greenhouse gases. As human society has –more or less– agreed on the need of CC mitigation, agriculture and its downstream process chains should safeguard that carbon fixed by biomass is re-emitted preferably as CO_2 instead of as more potent GHG. However, society needs to accept natural restrictions where agricultural activities are inevitably connected with GHG occurence (for instance N_2O from soils, or physiologically determined CH_4 from ruminant husbandry). Furthermore, it must accept natural variance and regional differences. Agriculture also provides cycles for potent climate effective gases. Hence, recognizing carbon as a resource that needs to be deployed most efficiently (instead of as impact on climate) might help to foster the transformation to a bio-based, low-carbon economy that could fulfil sustainability constraints by shaping sustainable, even though complex, biomass usage chains.

From the case studies and this dissertation, the following incomprehensive list presents aspects that might deserve further attention:

- Application of the *CUDe* approach to other biomass transformation pathways and evaluation of its transferability to food/feed or to other relevant material flows in agriculture (nitrogen, phosphorous)
- Integration of *CUDe* with other indicators, for example in a two-dimensional matrix together with an energy indicator
- Examination of possible existence of relevant VOC emissions from other agricultural commodities than SRC
- Identification of possible existence of relevant material flows associated with agriculture that might have not yet received attraction
- Increasing data availability, closing of data gaps
  ○ N_2O background emissions of different crops that might act as reference for bioenergy crops
  ○ Long-term measurements for SOC pool changes under SRC (including stock changes after returning to annual crop management at the plantation site)
  ○ Relevance and measurement methodology of CH_4 emissions from SRC storage
  ○ Alternative CO_2 flux data generation from biomass (citizen science; Fritz *et al.* 2016)
- Development of ILUC factors, other than for GHG (for example for biodiversity)
- Evaluation of communication alternatives for complex subjects (such as CC mitigation with bioenergy) in order to promote actions – using games, for example Tavoni *et al.* (2011)

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10 Abbreviations and Acronyms

AGB	Above-Ground Biomass
AR4,5	Fourth/Fifth Assessment Report (of IPCC)
ASEAN	Association of South East Asian Nations
BAU	Business As Usual
BGB	Below-Ground Biomass
BUE	Biomass Utilization Efficiency
C	Carbon
CC	Climate Change
CCS	Carbon Capture and Storage
CF	Conversion Factor
CH_4	Methane
CHP	Combined Heat and Power
CO_2	Carbon dioxide
CO_{2e}	Carbon dioxide equivalents
CSF	Carbon Stability Factor
CUDe	Carbon Utilization Degree
dLUC	direct Land Use Change
E_B (or EF_B)	Emission Factor (Bioenergy)
E_F (or EF_F)	Emission Factor (Fossil Energy)
EPS	Expanded Polystyrene
ETS	European Union Emission Trading Scheme
EU	European Union
FEA	Federal Environmental Agency (Umweltbundesamt, UBA)
FU	Functional Unit
g	Gram
GHG	Greenhouse Gas(es)
G	Giga (10^9)
GWP	Global Warming Potential
h	Hour
ha	Hectare
IEA	International Energy Agency
iLUC	indirect Land Use Change
IPCC	Intergovernmental Panel on Climate Change
IQR	Interquartile Range
K	Kelvin
K_2O	Potassium Oxide
kg	Kilogram
km	Kilometer
kWh	Kilowatt Hour
L	Insulation Layer Thickness
λ	Material-specific Thermal Conductivity
LC	Land Cover
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LU	Land Use
LUC	Land Use Change
LULUCF	Land Use, Land Use Change and Forestry
m/m^2/m^3	Meter/Square Meter/Cubic Meter

MC	Monte Carlo Analysis
MF_B	Mitigation Factor (Bioenergy)
MJ	Megajoule
MP_B	Mitigation Potential
N	Nitrogen
Mg	Megagram
n/a	Not available
N_2O	Nitrous Oxide
NF_3	Nitrogen trifluoride
NUE	Nitrogen Use Efficiency
P_2O_5	Phosphate
r/r_P	Ratio/Ratio as Percent
R	Rectangular
ϱ	Raw Density
RED	Renewable Energy Directive
SAR	Second Assessment Report (from IPCC)
SD	Standard Deviation
SE	Standard Error
SF_6	Sulphur hexafluoride
SOC	Soil Organic Carbon
SRC	Short Rotation Coppice
t	ton
THG	Treibhausgas(e)
TWh	Terawatt Hour
U	Specific Heat Transfer Coefficient
UNFCCC	United Nations Framework Convention on Climate Change
VOC	Volatile Organic Compounds
v/v	Volume/Volume
W	Watt
WBGU	Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen
w/w	Weight/Weight
yr^{-1}	Per Year
1G/2G/3G	First-/Second/Third-generation

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12 Supplements

12.1 Possible reasons for blurred sequestration reports under energy-crop plantations

(Extract from manuscript draft; Hansen, A. & S.Voß, unpublished)

12.1.1 Introduction

After LUC was raised as an issue of utmost importance in the performance analysis of biofuels (Searchinger et al., 2008), several authors gathered published data of C sequestration under bio-energy crop plantations (Don et al., 2011; 0.438±0.428 t C $ha^{-1}yr^{-1}$) or after LUC in general (Poeplau et al., 2011).

Investigating the original basic studies of the reviews, one can identify several possible reasons for the difference in the identified carbon sequestration (rates) of perennial bio-energy crop plantations, regardless of the species type (trees, e.g. poplar, willow, aspen; or perennial grasses, for example *Miscanthus* or switchgrass).

12.1.2 Possible reasons

12.1.2.1 Analyzed Compartment

Which volume of soil column was investigated? Were samples taken from the top soil only, did they include the organic surface layer, or were maybe also deeper profile horizons probed?

Carbon stocks as well as carbon sequestration, are distributed over the whole soil profile and changes might not be detected (or misinterpreted) if deeper layers are not investigated. Especially if long-term sequestration is aimed at, the deeper layers are important (Powlson et al., 2011). Kravchenko and Robertson (2011) analyzed the influence of the measurement depths on the significance of carbon sequestration. Also, Schmidt et al. (2011) point out that although samples are usually taken only in the topsoil, the deeper stored carbon might be essential. Schlesinger and Lichter (2001) reported that in short-term assessments with low assessment depths, effects could not be detected. Furthermore, the residue management, which is represented in the organic surface layer, provides (or withholds) those carbon stocks that can be transformed into longer-lasting soil carbon pools and should therefore be addressed as an important factor for carbon sequestration (see Blanco-Canqui, 2010; Sartori et al., 2006). Identical sample depths are pre-requisites for adequate comparisons between plots when results are reported in weight per hectare.

12.1.2.2 Status of the soil carbon pool

Can the soil at the experimental site already be judged to have arrived at a new carbon content equilibrium concerning historical land-use, climatic conditions and biomass input? Can the plantation be considered old enough to have an equilibrated SOC level?

At already carbon-stable sites, succeeding time-series measurements can be considered informative, whereas for plots that are still unbalanced the reference measurements should take place on adjacent plots under similar conditions regarding previous land-use, climatic conditions, and soil characteristics, as e.g. initial carbon stocks (Sanderman and Baldock, 2010). Otherwise, these underlying background variations in site properties may have overruled effects from the bio-energy crop cultivation. This is also important because of the hysteresis effect of SOC losses seeing that the C losses are happening faster than the carbon is being fixed in soils again.

Laganière et al. (2010) very often missed a key factor in their meta-analysis of afforestation studies on agricultural soils, namely, the validation of the basic premise for paired plots measurements, which is uniformity of site properties. Therefore, they suggest combining a paired plot sample design with chronosequences, or at best, using a retrospective design (time-series) and re-sample the same plots. We argue here with Sanderman and Baldock (2010) that this approach is only sufficient when soils have already reached their equilibrium. The chronosequence approach is not to be confused with a time-series design: the former takes samples of neighboring plots, which are assumed to be in different stages of similar development trajectories across multiple time-scales of the temporal dynamics of plant communities or soil development (see Walker et al., 2010 for a discussion of preconditions for this method). The latter takes samples of the same plot(s) repeatedly over a given period of time. This terminology is not used consistently in literature across the disciplines.

12.1.2.3 Different measurement methodologies

Besides the above mentioned sampling design with respect to soil carbon equilibrium, the alteration in soil bulk density also has an influence on the calculated sequestration rates. If the measurements are not corrected regarding the change of soil bulk density after the LUC has taken place, the sequestration might be misinterpreted (see Poeplau et al., 2011, p. 2417).

Further differences might be due to diverse experimental equipment (e.g. sieve diameters, chemicals) and methodology (drying temperatures, measurement dates (spring/autumn)).

Another approach to arrive at assumptions for carbon emissions due to LUC is to measure the carbon fluxes directly in the field. However, due to its laborious and costly intensity, this approach is taken rarely and usually the stock balancing method is followed. The application of stock and stock change factors, as they are promoted by the IPCC tier approach, was discussed by Sanderman and Baldock (2010). They argue that the underlying assumptions of a 20 year timescale for (a) reaching a C content equilibrium and (b) assuming the baseline C contents are at a steady-state might be sufficient for some soils but inappropriate for others. Additionally, carbon losses are probably proportional to the carbon stock but carbon gains are proportional to the carbon input to the C pool. This is not represented in the IPCC approach, where solely the linear equation C stock x stock change factor is followed (Sanderman and Baldock, 2010).

12.1.2.4 Experimental design

Was the experimental design sufficient to address the points mentioned above in a statistically significant way?

Kravchenko and Robertson (2011) point out that due to the natural variability of soils, it is essential to sample sufficient numbers of replicates to reduce the Type II error, that is to infer "no difference" between treatments (SRC and reference land-use), despite it exists. They found this to be an important reason for the lack of statistical differences in reported study results. Only an appropriate experimental design can increase the probability of detecting even substantial differences in SOC stocks. They also stress that analyses should be conducted for the different soil layers separately, as C stocks vary intensely among depth increments as well as between time-scales. Whole profile analyses should then be based on the incremental analyses. On the other hand, they also had to observe that studies do not always provide sufficient measures of variability.

12.1.2.5 Time aspects

Are the sequestration rates derived from long-term investigations or extrapolated from short-term experiments? How were timely effects accounted for (old and new carbon equilibrium, hysteresis effect of C sequestration, temporal emissions)?

It is still unclear how long it takes for a soil to reach its new equilibrium (Don et al., 2011; Powlson et al., 2011). This is partly due to the complexity of the dynamics of SOC pools (c.f. Schmidt et al., 2011). Positive effects of SRC, established on cropland, were found only after at least 12-15 years of plantation age (c.f. modeling results in Garten Jr et al., 2011), which is still a short period, considering that second generation crops (i.e. ligneous plants) usually are not cultivated annually but are long-term establishments on arable land. Nevertheless, as such long-term measurements are often not available, it is risky to extrapolate the initial rapid sequestration rates (Powlson et al., 2011). Linked to the equilibrium question is the uncertainty as to how long the additionally sequestered carbon will stay in the soil carbon pool, i.e. how effective the soil memory for carbon sequestration will be. Poeplau et al. (2011) found that cropland conversion might provide a C sink lasting longer than 100 years and therefore also propose to reconsider the IPCC time horizon of 20 years (Cherubini et al., 2011).

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12.2 Supplementary Data to Article [1] (Hansen *et al.* 2013) (6.1)

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Table 12.1: Diesel consumption for the cultivation of SRC for a plantation with a 4-year harvest cycle and a standing time of 16 years

Farming operations	Diesel fuel per operation [L ha^{-1}]
Establishment of the plantation	
- Commuting to soil probing	0.1
- Pesticide application in autumn	1.0
- Ploughing	22.9
- Harrowing	5.8
- Transport and Planting of saplings	0.4
- Pesticide application in spring	1.0
- Hoeing	8.0
Harvest (four times within plantation standing time)	
- Harvesting	120
- Commuting to soil probing	0.1
Phosphorous/Potassium fertilizer application after harvest if necessary (four times within plantation standing time)	1.3
Recultivation	23
Total diesel consumption for farming operations per complete standing time	547.8

Data according to [35]: KTBL, editor. Energiepflanzen - Daten für die Planung des Energiepflanzenanbaus. Darmstadt: KTBL Kuratorium für Technik und Bauwesen in der Landwirtschaft (2006)